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

Action Agencies:	The United States Navy and Permits and Conservation Division, Office of Protected Resources, National Marine Fisheries Service, National Oceanic and Atmospheric Administration
Activity Considered:	(1) United States Navy's Surveillance Towed Array Sensor System Low Frequency Active Sonar Routine Training, Testing, and Military Operations (2) National Oceanic and Atmospheric Administration's National Marine Fisheries Service, Office of Protected Resources, Permits and Conservation Division's Issuance of Four Letters of Authorization for the U.S. Navy to "Take" Marine Mammals Incidental to Surveillance Towed Array Sensor System Low Frequency Active Sonar Routine Training, Testing, and Military Operations in Areas of the Pacific Ocean for the Period of August 15, 2016 through August 14, 2017 pursuant to the Five-Year Marine Mammal Protection Act Regulations
Consultation Conducted By:	Endangered Species Act Interagency Cooperation Division, Office of Protected Resources, National Marine Fisheries Service, National Oceanic and Atmospheric Administration

Approved:

for
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1 INTRODUCTION

The Endangered Species Act (ESA) of 1973, as amended (16 U.S.C. 1531 et seq.) establishes a national program for conserving threatened and endangered species of fish, wildlife, plants, and the habitat they depend on. Section 7(a)(2) of the ESA requires Federal agencies to insure that the actions they authorize, fund, or carry out are not likely to jeopardize the continued existence of endangered or threatened species or adversely modify or destroy their designated critical habitat. When a Federal agency's action "may affect" a protected species, that agency is required to consult formally with National Marine Fisheries Service (NMFS) or the U.S. Fish and Wildlife Service (USFWS), depending upon the endangered species, threatened species, or designated critical habitat that may be affected by the action (50 CFR §402.14(a)).

Federal agencies shall confer with the NMFS or USFWS on any action which is likely to jeopardize the continued existence of any proposed species or result in the destruction or adverse modification of proposed critical habitat (50 CFR §402.10). If requested by the Federal agency and deemed appropriate, the conference may be conducted in accordance with the procedures for formal consultation in §402.14.

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

The United States (U.S.) Navy submitted a request for formal ESA consultation based on their determination that multiple ESA-listed species under NMFS' jurisdiction may be affected by the proposed operation of the SURTASS LFA sonar onboard four U.S. Navy vessels. NMFS' Permits and Conservation Division submitted a request for formal consultation based on their determination that the issuance of four Letters of Authorization (LOAs) pursuant to Federal regulations under the Marine Mammal Protection Act of 1972, as amended (MMPA; 16 U.S.C. 1361 et seq.), related to the U.S. Navy's use of SURTASS LFA sonar may affect several ESA-listed species.

For the actions described in this document, the action agencies are: (1) the U.S. Navy, which proposes to operate the SURTASS LFA sonar onboard four Tactical-Auxiliary General Ocean Surveillance (T-AGOS) vessels; and (2) NMFS Permits and Conservation Division, which proposes to issue four Letters of Authorization that would allow the U.S. Navy to "take" marine mammals incidental to those SURTASS LFA sonar activities. The consulting agency for these proposals is NMFS Office of Protected Resources, Endangered Species Act Interagency Cooperation Division.

The biological opinion (opinion), conference report, and incidental take statement were prepared by NMFS Endangered Species Act Interagency Cooperation Division in accordance with section 7(b) of the ESA and implementing regulations at 50 CFR §402. This document represents NMFS' final opinion on the effects of these actions on endangered and threatened species, or proposed species, and critical habitat that has been designated, or is proposed for designation, 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

We issued the first five-year biological opinion on U.S. Navy SURTASS LFA sonar activities in 2002 (NMFS 2002) and NMFS Permits and Conservation Division's promulgation of five-year regulations valid from 2002 through 2007 under the MMPA following the completion of a Final Overseas Environmental Impact Statement and Environmental Impact Statement (FOEIS/EIS) (Navy 2001a; Navy 2001b; Navy 2001c) pursuant to the National Environmental Policy Act of 1969 (NEPA; 42 U.S.C. 4321 et seq. The MMPA regulations allowed for the issuance of annual LOAs to the U.S. Navy that authorized the take of marine mammals incidental to the conduct of SURTASS LFA sonar operations from the R/V *Cory Chouest* and the USNS *IMPECCABLE*. Annual biological opinions were subsequently written on the issuance of each annual LOA pursuant to the MMPA regulations. Each of the biological opinions completed from 2002 through 2007 concluded that the proposed actions would not jeopardize any ESA-listed species and would not result in the destruction or adverse modification of any designated critical habitat under NMFS jurisdiction (NMFS 2002; NMFS 2003; NMFS 2004b; NMFS 2005c; NMFS 2007c).

On August 13, 2007, we completed a biological opinion on a second five-year period of SURTASS LFA sonar activities and a corresponding MMPA rule that was valid from August 2007 through August 2012, following the necessary analysis under NEPA (Navy 2007). In the 2007 biological opinion, we evaluated the potential effects of the U.S. Navy's use of SURTASS LFA sonar on up to four vessels and NMFS Permits and Conservation Division's promulgation of regulations under the MMPA (72 FR 46846, August 21, 2007) for the "take" of marine mammals over a five year period. This rule was again followed by the issuance of annual LOAs and related biological opinions. In the biological opinions completed from 2007 through 2011, we concluded that the proposed actions would not jeopardize any ESA-listed species and would not result in the destruction or adverse modification of any critical habitat under NMFS jurisdiction (NMFS 2007c; NMFS 2008a; NMFS 2009b; NMFS 2010a; NMFS 2011a).

Following the U.S. Navy's NEPA analysis, SURTASS LFA sonar 2012 Supplemental Environmental Impact Study / Supplemental Overseas EIS (SEIS/SOEIS) (Navy 2012a), we completed a biological opinion on the third five-year period of SURTASS LFA sonar operation and the related MMPA rule on August 13, 2012, which is valid from August 15, 2012 through August 14, 2017 (NMFS 2012b). These regulations (50 CFR § 218 Subpart X) are pursuant to section 101(a)(5)(A) of the MMPA and again make it possible for NMFS Permits and Conservation Division to issue annual LOAs that allow the U.S. Navy to take marine mammals incidental to the U.S. Navy's operations of SURTASS LFA sonar.

In summary, the MMPA regulations require the U.S. Navy to (1) conduct the activities in a manner that minimizes, to the greatest extent practicable, any adverse impacts on marine mammals and their habitat, (2) only take marine mammals by the method of take indicated in the regulation and LOAs, (3) maintain a running calculation/estimation of takes of each species over the effective period of these regulations, (4) ensure that no more than 12 percent of any marine mammal stock would be taken by Level B harassment[1] annually over the course of this five-year regulation, regardless of the number of LFA sonar vessels operating, (5) limit Level A harassment as defined by the MMPA to no more than six mysticetes (total), of any of the species listed in 50 CFR § 218.230(b)(1) over the course of the five-year regulations, (6) limit Level A harassment as defined by the MMPA to no more than 25 odontocetes (total), of any of the species listed in 50 CFR § 218.230(b)(2) over the course of the five-year regulations, and (7) limit Level A harassment as defined by the MMPA to no more than 25 pinnipeds (total), of any of the species listed in 50 CFR § 218.230(b)(3) over the course of the five-year regulations.

This rule was again followed by the issuance of annual LOAs and related biological opinions. On August 15, 2012, NMFS issued its biological opinion on the first LOAs on the effects of the U.S. Navy's use of SURTASS LFA sonar on up to four vessels and NMFS Permits and Conservation Division's proposed issuance of LOAs that would be valid from August 15, 2012 through August 14, 2013. In that biological opinion we concluded that SURTASS LFA activities were not likely to jeopardize any ESA-listed species and were not likely to result in the destruction or adverse modification of any critical habitat under NMFS jurisdiction (NMFS 2012a). On August 15, 2013, NMFS issued a biological opinion on the effects of the U.S. Navy's use of SURTASS LFA sonar on up to four vessels and NMFS Permits and Conservation Division's issuance of LOAs that would be valid from August 15, 2013 through August 14, 2014. In that biological opinion we also concluded that SURTASS LFA activities were not likely to jeopardize any ESA-listed species and were not likely to result in the destruction or adverse modification of any critical habitat under NMFS jurisdiction. On August 13, 2014, NMFS issued a biological opinion on the effects of the U.S. Navy's use of SURTASS LFA sonar on up to four vessels and NMFS Permits and Conservation Division's issuance of LOAs that are valid from August 15, 2014 through August 14, 2015. In that biological opinion we concluded that SURTASS LFA activities were not likely to jeopardize any ESA-listed species and were not likely to result in the destruction or adverse modification of any critical habitat under NMFS jurisdiction. Most

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[1] The MMPA defines "harassment" as "any act of pursuit, torment, or annoyance which has the potential to injure a marine mammal or marine mammal stock in the wild or has the potential to disturb a marine mammal or marine mammal stock in the wild by causing disruption of behavioral patterns, including, but not limited to, migration, breathing, nursing, breeding, feeding, or sheltering" [16 U.S.C. 1362(18) (A)]. The NMFS has not adopted a regulatory definition of harassment under the ESA.

recently, on August 4, 2015, NMFS issued a biological opinion on the effects of the U.S. Navy's use of SURTASS LFA sonar on up to four vessels and NMFS Permits and Conservation Division's issuance of LOAs that would be valid from August 15, 2015 through August 14, 2016. In that biological opinion we also concluded that SURTASS LFA activities were not likely to jeopardize any ESA-listed species and were not likely to result in the destruction or adverse modification of any critical habitat under NMFS jurisdiction.

1.2 Consultation History

This opinion and conference report is based on information provided by the U.S. Navy and NMFS Permits Division as well as monitoring reports and other sources of information. Our communication with the U.S. Navy and Permits Division regarding this consultation is summarized as follows:

On April 11, 2016, we received a request from the U.S. Navy to initiate formal ESA section 7 consultation on SURTASS LFA sonar activities that would occur from August 15, 2016 through August 14, 2017.

On June 13, 2016, we received a request from NMFS Permits and Conservation Division for formal ESA section 7 consultation on their proposal to issue four annual LOAs pursuant to the MMPA on the U.S. Navy's use of SURTASS LFA sonar aboard four vessels. The reinitiation package included draft LOAs.

On June 28, 2016, we initiated consultation and subsequently prepared this opinion, conference report, and incidental take statement in accordance with section 7(b) of the ESA of 1973, as amended (16 U.S.C. 1531, *et seq*.), and implementing regulations at 50 CFR § 402.

2 DESCRIPTION OF THE PROPOSED ACTION

"Action" means all activities or programs of any kind authorized, funded, or carried out, in whole or in part, by Federal agencies. Interrelated actions are those that are part of a larger action and depend on the larger action for their justification. Interdependent actions are those that have no independent utility apart from the action under consideration. This opinion and conference report considers two related actions proposed by the U.S. Navy and NMFS Permits and Conservation Division. The U.S. Navy proposes SURTASS LFA sonar routine training, testing, and military operations on up to four vessels. NMFS Permits and Conservation Division proposes to issue four LOAs to the U.S. Navy governing the take and importation of marine mammals (50 CFR § 218 Subpart X). The four LOAs would authorize the U.S. Navy to take marine mammals incidental to their routine training, testing, and military operations of SURTASS LFA sonar onboard four vessels. These LOAs would be effective for one year from August 15, 2016 to August 14, 2017. The purpose of the U.S. Navy's proposed action is to meet the United States' need for an improved ability to detect quieter and harder-to-find foreign submarines at long range to provide U.S. forces with adequate time to respond to potential submarine threats (Navy 2012a). This opinion and conference report supercedes the 2015 opinion.

The approach of addressing interdependent actions is consistent with Congress' intent that we coordinate and integrate the decision-making process under MMPA and ESA to the maximum extent practicable, so this opinion and conference report analyzes the routine training, testing, and military operational activities during the time and in the geographic area covered by the MMPA regulations, which are limited to "periods of not more than five consecutive years." 16 U.S.C. 1371(a)(5)(A)(i). Further, NMFS has determined to structure this consultation in this way to ensure that the effects of reasonably anticipated routine training, testing, and military operational activities may be analyzed close in time to their occurrence.

NMFS recognizes that while U.S. Navy SURTASS LFA routine training, testing, and military operational requirements change over time in response to global or geopolitical events and other factors, the general types of activities addressed by this consultation are expected to continue into the reasonably foreseeable future, along with the associated impacts. Therefore, as part of our effects analysis, we assumed that the activities proposed for the period of August 15, 2016, through August 14, 2017, and the remaining period of the five-year (2012 to 2017) MMPA rule would continue into the reasonably foreseeable future at levels similar to that assessed in this opinion and conference report, and we considered the direct and indirect effects of those assumed future activities, together with the effects of all interrelated and interdependent actions. This approach addresses the court decision in Intertribal Sinkyone Wilderness Council v. National Marine Fisheries Service et al., No. 1:12-cv-00420-NJV (N.D. Cal. Sept. 25, 2013), although we may consider a different approach in future actions.

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 U.S. Navy applies for a new MMPA incidental take authorization for SURTASS LFA upon expiration of the five-year regulations considered in this opinion and conference report. The U.S. Navy would also need to initiate a new ESA consultation at that time.

2.1 U.S. Navy Proposed Activities

The U.S. Navy proposes to use SURTASS LFA sonar systems (LFA sonar or compact LFA [CLFA] sonar[2]) on a maximum of four United States Naval Ships during routine training and testing as well as during military operations, including 12 mission areas of the western and central North Pacific Ocean and three mission areas in the Indian Ocean from August 15, 2016 through August 14, 2017. The vessels are: United States Naval Ship (USNS) *IMPECCABLE* (T-

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[2] Hereafter, LFA sonar refers to both the LFA and CLFA sonar systems.

AGOS 23), USNS *ABLE* (T-AGOS 20), USNS *EFFECTIVE* (T-AGOS 21), and USNS *VICTORIOUS* (T-AGOS 19). No more than four LFA sonar systems are expected to be in use during this period. Over the 2016 to 2017 effective period, the U.S. Navy plans to employ SURTASS LFA sonar systems onboard four vessels during no more than 20 total missions.

2.1.1 Deployment

The U.S. Navy proposes an estimated combined total of 20 active sonar missions for four SURTASS LFA vessels in the following operational areas:

- Up to 16 missions in the western North Pacific Ocean, which includes the following mission areas: east of Japan; North Philippine Sea; west Philippine Sea; offshore Guam; Sea of Japan; East China Sea; South China Sea; offshore Japan (25 to 40° North and 10 to 25° North), and northeast of Japan.

- Up to two missions in the central North Pacific Ocean, which includes the northern and southern Hawaii mission areas.

- Up to two missions in the Indian Ocean, which includes the Arabian Sea, Andaman Sea, and northwest of Australia mission areas.

Each vessel is expected to spend approximately 54 days in transit between ports and operation areas as well as up to 240 days performing active sonar operations. Between missions, an estimated total of 71 days per year will be spent in port for upkeep and repair to maintain both the material condition of the vessel, its systems, and the morale of the crew. The maximum number of actual sonar transmission hours per vessel for the period will not exceed 432 hours. Therefore, the total number of active sonar hours for all vessels will not exceed 1,728 hours over the one-year period.

2.1.2 Active Sonar Components

The SURTASS LFA sonar systems are long-range sensors that operate in the low frequency band (between 100 and 500 Hertz [Hz]). SONAR is an acronym for SOund NAvigation and Ranging, and its definition includes any system that uses underwater sound, or acoustics, for observations, monitoring, and communications. Sonar systems are used for many purposes, ranging from commercial off-the-shelf “fish finders” to military anti-submarine warfare (ASW) systems for detection and classification of submarines. More information on the LFA sonar systems can be found in the SURTASS LFA sonar 2012 Supplemental Environmental Impact Study/Supplemental Overseas EIS (SEIS/SOEIS) (Navy 2012a).

The characteristics and operating features of the active acoustic component of LFA sonar are:

- The acoustic source is a vertical line array (VLA) of up to 18 source projectors suspended beneath the vessel (Figure 1). The LFA sonar’s transmitted beam is omnidirectional (360 degrees) in the horizontal, with a narrow vertical beamwidth that can be steered above or below the horizontal.

- The source frequency is between 100 and 500 Hz. A variety of signal types can be used, including continuous wave (CW) and frequency-modulated (FM) signals.
- The source level (SL) of an individual source projector of the SURTASS LFA sonar array is approximately 215 dB re: 1 micro Pascal (µPa) at 1 meter (m) root mean squared (rms) or less. As measured by sound pressure level (SPL), the sound field of the array can never be higher than the SL of an individual source projector.

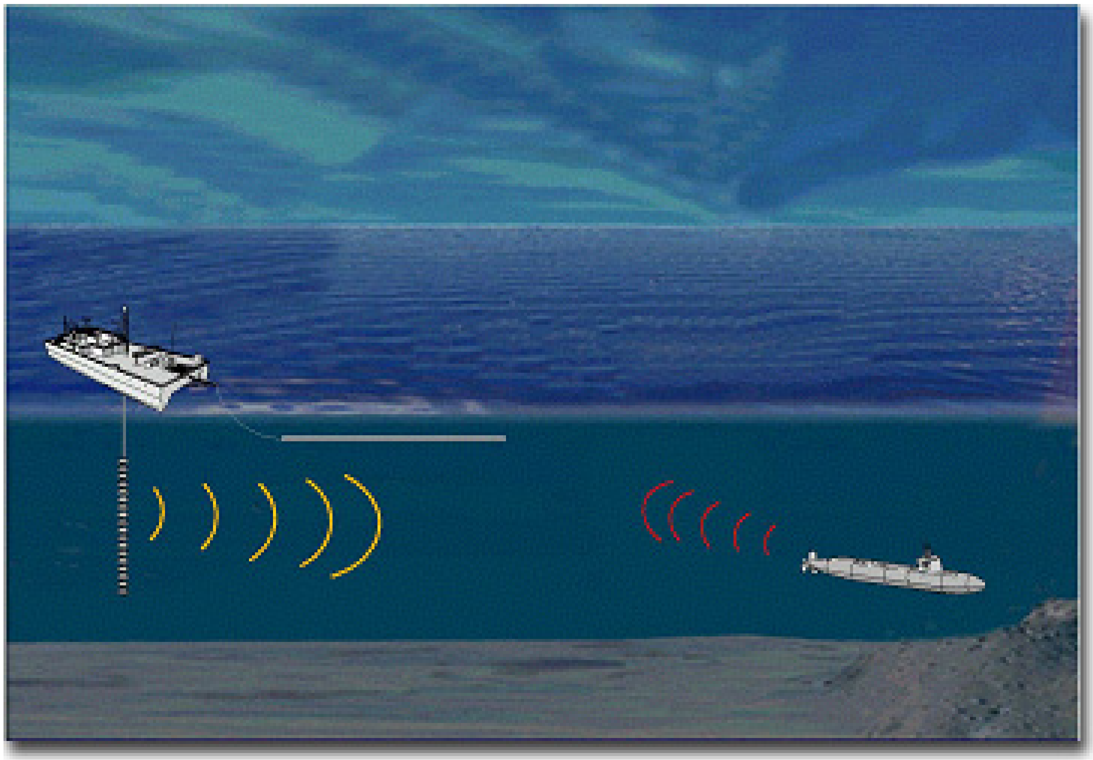

Figure 1. SURTASS LFA sonar systems showing the active (source array) and passive (receive array) components.

2.1.3 Passive Sonar Components

The SURTASS passive, or listening, part of the system detects returning echoes from submerged objects, such as threat submarines, through the use of hydrophones. These devices transform mechanical energy (received acoustic sound waves) to an electrical signal that can be analyzed by the processing system of the sonar. Advances in passive acoustic technology have led to the development of the SURTASS Twin-line (TL-29A) horizontal line array, a shallow water variant of the single line SURTASS system. TL-29A consists of a "Y" shaped array with two apertures. The array is approximately 1/5th the length of a standard SURTASS array, or approximately 305 m (1,000 ft) long. The TL-29A delivers enhanced capabilities, such as its ability to be towed in shallow water environments in the littoral zone, to provide significant directional noise rejection, and to resolve bearing ambiguities without having to change vessel course. The passive capabilities of all SURTASS LFA sonar vessels have been upgraded with the installation of the TL-29A array.

The SURTASS LFA sonar vessel typically maintains a speed of at least 5.6 kilometers per hour (kph) (3 knots) through the water when towing the horizontal line array. The return signals received by the SURTASS array, which are usually below background or ambient noise level, are then processed and evaluated to identify and classify potential underwater threats.

2.1.4 Mitigation and Monitoring

The following mitigation and mitigation monitoring measures are required when SURTASS LFA sonar is transmitting to reduce the potential for injury to marine animals. The LFA mitigation zone covers the ocean volume ensonified to received levels of 180 dB re: 1 µPa (rms) SPL and greater by the SURTASS LFA sonar transmit array. Prior to commencing and during SURTASS LFA sonar transmissions, the U.S. Navy will determine the propagation of LFA sonar signals in the ocean and the distance from the SURTASS LFA sonar source to the 180-dB re: 1 µPa isopleth (i.e., the LFA sonar mitigation zone). As an added measure, NMFS requires a ‘‘buffer zone’’ that extends an additional one km (0.54 nmi) beyond the 180-dB LFA sonar mitigation zone.

2.1.4.1 Visual Monitoring

Visual monitoring includes daytime observations from the SURTASS LFA sonar vessels for marine mammals and sea turtles by personnel trained to detect and identify marine mammals and sea turtles. Visual monitoring begins 30 minutes before sunrise or 30 minutes before the first SURTASS LFA sonar transmission. Monitoring continues for at least 15 minutes after completion of the SURTASS LFA sonar transmission exercise or until 30 minutes after sunset. The objective of these visual observations is to ensure that no marine mammals or sea turtles approach close enough to enter the LFA sonar mitigation and buffer zones and to maintain a record of observed marine mammals and/or sea turtles.

Visual observations are made by designated ship personnel trained in detecting and identifying marine mammals and sea turtles. The trained observers maintain a topside watch and marine mammal/sea turtle observation log during any active SURTASS LFA sonar transmissions. The number and identification of marine mammals/sea turtles sighted, date/time of sighting, bearing and range from vessel, as well as any unusual behavior, is entered into the log.

If a potentially affected marine mammal or sea turtle is sighted outside of the LFA sonar mitigation and buffer zones, the observer notifies the Officer in Charge (OIC) of the military crew (MILCREW). The MILCREW OIC then notifies the sonar operator for the high frequency /marine mammal monitoring (HF/M3) sonar (discussed in more detail below) to determine the range and projected track of the animal. If it is determined that the animal will travel into the

LFA sonar mitigation and buffer zones, the MILCREW OIC orders the delay or suspension of SURTASS LFA sonar transmissions when the animal is observed entering the LFA sonar mitigation and buffer zone. If the animal is visually observed anywhere within the LFA sonar mitigation and buffer zone, the MILCREW OIC orders the immediate suspension of SURTASS LFA sonar transmissions. The observer continues visual monitoring and recording until the animal is no longer seen. All sightings are recorded in the sighting log.

2.1.4.2 Passive Acoustic Monitoring

Passive acoustic monitoring will be conducted using the passive (low frequency) SURTASS towed horizontal line array to listen for sounds generated by marine mammals as an indicator of their presence. If a received sound is estimated to be from a marine mammal within proximity of the SURTASS LFA sonar vessel, the monitoring technician notifies the MILCREW, who in turn alerts the HF/M3 sonar operator (discussed in more detail below) and visual observers (during daylight hours) to the presence of a marine mammal. If passive acoustic monitoring detects a marine mammal prior to or during sonar transmissions, the MILCREW orders a delay or suspension of SURTASS LFA sonar transmissions when the HF/M3 sonar and/or visual observers indicate that the animal enters the LFA sonar mitigation and buffer zones. All contacts are recorded in the log.

2.1.4.3 Active Acoustic Monitoring

Active acoustic monitoring uses the HF/M3 sonar (Figure 2), which is a U.S. Navy-developed, enhanced HF commercial sonar designed to detect, locate, and track marine mammals (and possibly sea turtles, sturgeon, and scalloped hammerhead sharks), that may pass close enough to the SURTASS LFA sonar’s transmitter array to enter the LFA sonar mitigation and buffer zones.

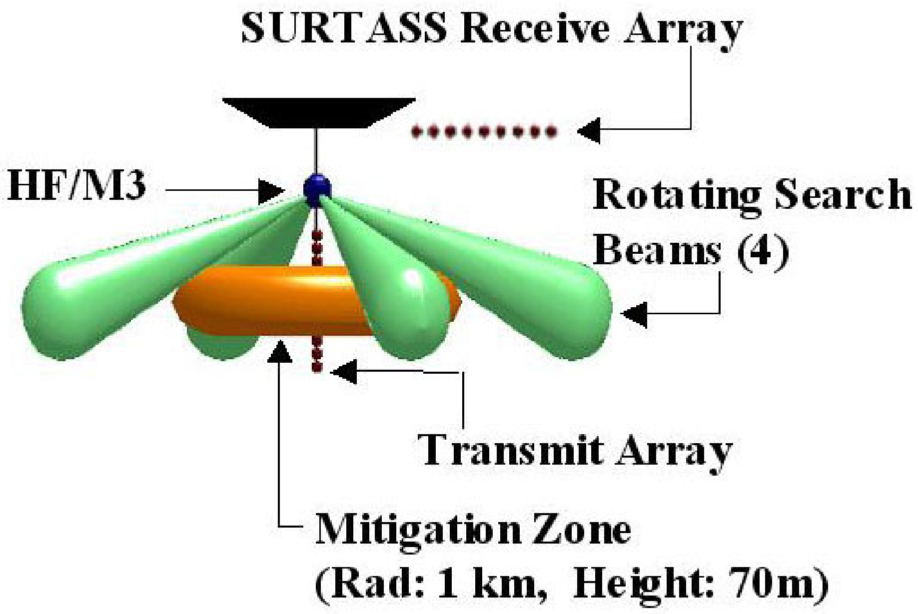

Figure 3. High Frequency / Marine Mammal Monitoring (HF/M3) Sonar

HF/M3 acoustic monitoring begins 30 minutes before the first SURTASS LFA sonar transmission of a given mission is scheduled to commence and continues until sonar transmissions are terminated. Prior to full-power sonar operations, the power level of the HF/M3 sonar is ramped up over a period of five minutes from the source level of 180 dB re: 1 µPa at 1 m (rms) in ten-dB increments until full power (if required) is attained to ensure that there are no inadvertent exposures of local animals to received levels greater than or equal to180 dB re: 1 µPa at 1 m (rms) from the HF/M3 sonar.

If the HF/M3 sonar detects a contact outside the LFA sonar mitigation and buffer zones, the HF/M3 sonar operator determines the range and projected track of the animal. If the operator determines that the animal will pass within the LFA sonar mitigation and buffer zones, the sonar operator notifies the MILCREW OIC. The MILCREW OIC then orders the delay or suspension of LFA sonar transmissions when the animal is predicted to enter the LFA sonar mitigation and buffer zones. Also, if a marine mammal or sea turtle is detected by the HF/M3 sonar within the LFA sonar mitigation and buffer zone, the operator notifies the MILCREW OIC, who orders the immediate delay or suspension of transmissions. All contacts are recorded in the log.

2.1.4.4 Resumption of SURTASS LFA Sonar Transmissions

The U.S. Navy may resume/commence SURTASS LFA sonar transmissions 15 minutes after 1) all marine mammals have left the area of the LFA sonar mitigation zone and the one-km buffer zone; and/or 2) there is no further detection of any marine mammal within the LFA sonar mitigation zone plus the one-km buffer zone as determined by the passive or active acoustic or visual monitoring protocols.

2.1.5 Mitigation

Mitigation, as defined by the U.S. Council on Environmental Quality (CEQ), includes measures undertaken to minimize impacts by limiting the degree or magnitude of a proposed action and its implementation. The objectives of the mitigation and monitoring measures presented for use when SURTASS LFA sonar is transmitting are to effect the least practicable adverse impact on marine mammal species or stocks and to reduce the likelihood of adverse effects to ESA-listed marine species or adverse effects to their designated critical habitats. Under the terms and conditions of current and past biological opinions and incidental take statements, the U.S. Navy is required to carry out all mitigation and monitoring requirements contained in the LOAs issued under section 101(a)(5)(A) of the MMPA. Summaries of the results of these requirements are documented in quarterly, annual, and comprehensive reports.

2.1.5.1 HF/M3 Sonar

The U.S. Navy will conduct active acoustic monitoring (through the use of HF/M3 sonar) to detect, locate, and track marine mammals (and possibly sea turtles, sturgeon, and scalloped hammerhead sharks) that pass close enough to the SURTASS LFA sonar array to exceed the 180-dB mitigation criterion. This HF/M3 sonar operates with a similar power level, signal type, and frequency as HF “fish finder” type sonars used worldwide by both commercial and recreational fishermen. Due to the depth of the LFA VLA, the HF/M3 sonar system was not designed to detect marine mammals or sea turtles at or near the surface in proximity to the SURTASS LFA vessel.

The HF/M3 sonar is located near the top of the SURTASS LFA sonar vertical line array (Figure 4), about 100 m below the sea surface. The general characteristics of the HF/M3 sonar are:

- Frequency: 30 to 40 kHz;
- Bandwidth: variable (1.5 to 6 kHz nominal);
- Duty Cycle: three to four percent (nominal);
- Nominal Source Level: 220 dB re: 1 µPa at 1 m;
- Pulse Length: variable (10 to 40 msec nominal);
- Pulse Repetition Rate: set by maximum search range (3 to 4 seconds nominal);
- Source Ramp-Up: five-minute period;
- Detection Volume: four equally spaced swept 8º (horizontal) x 10º (vertical) beams making up a 10º (vertical) sector sweep through full 360º (horizontal) around the source

(i.e., omnidirectional in the horizontal, 10° vertical beamwidth); nominal time for full 360° sweep 45 to 60 seconds;

- Maximum Detection Range: nominally two km (1.08 nmi);
- Operational Depth Capability: compatible with maximum deployed depth of SURTASS LFA sonar source array;
- Vertical Steering: ±10°; and
- Receiver Gain: 23 dB (nominal versus omnidirectional noise).

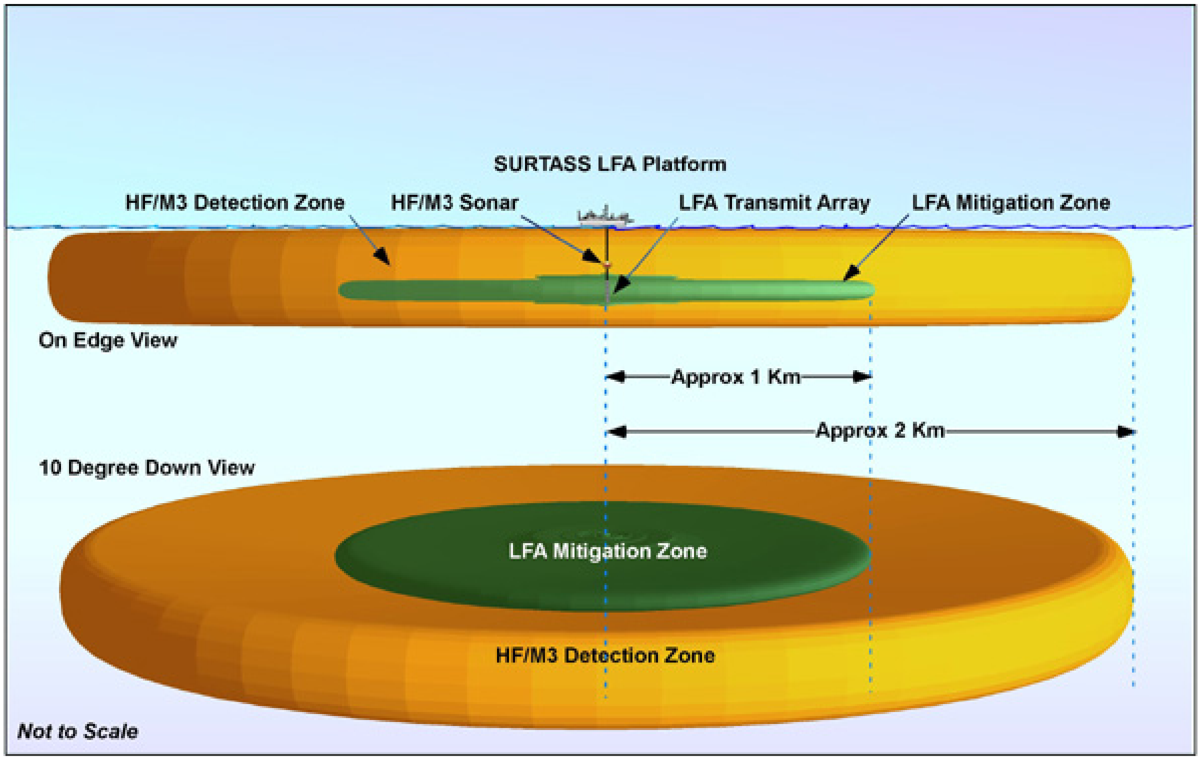

Figure 4. High frequency/marine mammal mitigation (HF/M3) sonar system detection range and low frequency active mitigation zone.

The HF/M3 sonar will operate 30 minutes prior to commencement of LFA sonar transmissions and continuously while the SURTASS LFA sonar is active. Detection of a marine animal by the HF/M3 sonar automatically triggers an alert to the OIC MILCREW, who has the HF/M3 tracking team immediately evaluate the detection.

Analysis and testing of the HF/M3 sonar operating capabilities indicates that this system substantially increases the probability of detecting marine mammals within the LFA sonar mitigation and buffer zones and provides an excellent monitoring capability (particularly for medium to large marine mammals) beyond the LFA sonar mitigation and buffer zones, out to approximately two km (1.08 nmi) (Ellison and Stein 1999). In fact, the U.S. Navy's tests have shown that the HF/M3 system is nearly 100 percent effective using multiple pings in detecting marine mammals of any size.

The HF/M3 system also increases the likelihood of detecting marine turtles, sturgeon, and scalloped hammerhead sharks. Because the HF/M3 sonar is positioned at the top of the LFA vertical array, sea turtles would first have to swim through the one-km HF/M3 sonar detection zone (i.e., the very same one-km buffer zone for marine mammals) where detection is highly likely, before entering the LFA sonar mitigation zone.

2.1.5.2 Coastal and Dive Site Restrictions

Based on the analyses presented in SURTASS LFA sonar NEPA documents (Navy 2001b; Navy 2001c; Navy 2007; Navy 2012a), NMFS and the U.S. Navy have developed geographic restrictions for the SURTASS LFA sonar sound field. During SURTASS LFA sonar operations, the sound field produced by the LFA sonar will not exceed 180 dB re: 1 µPa (rms) sound pressure level within 22 km (12 nmi) of any coastline (including offshore islands). This geographic restriction is known as the “coastal standoff range” for use of SURTASS LFA sonar. Additionally, the U.S. Navy will operate SURTASS LFA sonar so that the sound field will not exceed received levels of 145 dB re: 1 µPa (rms) within known recreational and commercial dive sites.

2.1.5.3 Offshore Biologically Important Areas (OBIAs) for Marine Mammals

For the use of SURTASS LFA sonar, OBIAs are defined as those areas of the world’s oceans outside of the geographic stand-off range of any coastline (i.e., 22 km [12 nmi]) with: (1) high densities of marine mammals; (2) known/defined breeding/calving grounds, foraging grounds, migration routes; or (3) small, distinct populations of marine mammals with limited distributions.

The 2012 NMFS MMPA Rule for the operation of SURTASS LFA sonar requires the U.S. Navy to ensure that the SURTASS LFA sonar sound field does not exceed 180 dB re: 1 µPa (rms) at least 1 km (0.54 nmi) seaward of the outer perimeter of the OBIA (50 CFR § 218 Subpart 234).

Under the 2012 MMPA regulations, 22 OBIAs have been designated for SURTASS LFA sonar. A detailed description of the OBIA derivation process, assessment conclusions, and specific OBIA boundary coordinates may be found in the Final SEIS/SOEIS for SURTASS LFA sonar (Navy 2012a) and in the final rulemaking for SURTASS LFA sonar. However, all of the OBIAs will be outside of the U.S. Navy’s proposed operational areas during the effective period of the proposed LOAs for 2016 to 2017 except the Hawaiian Islands Humpback Whale National Marine Sanctuary—Penguin Bank OBIA located in the North-central Pacific Ocean.

2.2 NMFS Permits and Conservation Division Proposed Activities (2016 to 2017)

NMFS Permits and Conservation Division proposes to issue four LOAs (one for each SURTASS LFA sonar vessel) pursuant to the MMPA implementing regulations and the regulations governing taking of marine mammals incidental to SURTASS LFA sonar activities that would authorize the incidental “take” of marine mammals by the U.S. Navy. The LOAs will include requirements for monitoring, mitigation, reporting, geographical restrictions, and long-term monitoring for SURTASS LFA sonar operations. The MMPA specifically limits the NMFS authority to allow for incidental, but not intentional takes, during periods of not more than five consecutive years (MMPA 101 (a)(5)(A)(i)). Each LOA is valid from August 15, 2016 through August 14, 2017 pursuant to MMPA regulations at 50 CFR § 218 Subpart X. The LOAs will allow the U.S. Navy to incidentally, but not intentionally, take marine mammals by harassment within the action area.

2.2.1 Authorization of Incidental Take of Marine Mammals

The Holder of the Authorization (i.e., U.S. Navy) must maintain a running calculation/estimation of takes of each stock over the effective period of these regulations. The take, by Level B harassment, that occurs during the year covered by the Authorization may not exceed 12 percent of any marine mammal stock (Table 1. to Table 11.).

Table 1. Estimated Level B Harassment Authorized for Mission Area 1, East of Japan

East of Japan—Mission Area 1 1.5 Missions	
Species	**Take Estimates Level B harassment**
Blue whale	2
Fin whale	3
Sei whale	17
Humpback whale	6
North Pacific right whale[1]	1
Sperm whale	20

[1] Although North Pacific right whales potentially occur in the East of Japan mission area in the fall through spring seasons, because the U.S. Navy modeled and presented take estimates for the summer season in this mission area when the North Pacific right whale does not occur, no take estimate was provided for this species. However, because North Pacific right whales are known to occur in low densities in the waters of this mission area during seasons that were not modeled (i.e., fall through spring), we estimate one right whale take will occur in this mission area.

Table 2. Estimated Level B Harassment Authorized for Mission Area 2, North Philippine Sea

North Philippine Sea—Mission Area 2 3 Missions	
Species	**Take Estimates Level B harassment**
Blue whale	2
Fin whale	11
Humpback whale	45
North Pacific right whale	2
Sperm whale	48

Table 3. Estimated Level B Harassment Authorized for Mission Area 3, West Philippine Sea

West Philippine Sea—Mission Area 3 3 Missions	
Species	**Take Estimates Level B harassment**
Blue whale	3
Fin whale	11
Humpback whale	67
Sperm whale	46

Table 4. Estimated Level B Harassment Authorized for Mission Area 4, Offshore Guam

Offshore Guam—Mission Area 4 3 Missions	
Species	**Take Estimates Level B harassment**
Blue whale	2
Fin whale	2
Sei whale	6
Humpback whale	11
Sperm whale	36

Table 5. Estimated Level B Harassment Authorized for Mission Area 5, Sea of Japan

Sea of Japan—Mission Area 5 1 Mission	
Species	**Take Estimates Level B harassment**
Fin whale	31
North Pacific right whale	0
Western North Pacific gray whale	1
Sperm whale	31
Spotted seal – Southern DPS	1

Table 6. Estimated Level B Harassment Authorized for Mission Area 6, East China Sea

East China Sea—Mission Area 6 1 Mission	
Species	**Take Estimates Level B harassment**
Fin whale	4

North Pacific right whale[1]	1
Western North Pacific gray whale[1]	1
Sperm whale	15
Spotted seal – Southern DPS	1

[1] Although North Pacific right and western North Pacific gray whales potentially occur in the East China Sea mission area in the fall through spring seasons, because the U.S. Navy modeled and presented take estimates for the summer season in this mission area when neither species occurs, the U.S. Navy did not provide take estimates for either the North Pacific right and western North Pacific gray whales in the East China Sea mission area. However, because these species are known to occur in low densities in the waters of this mission area during seasons that were not modeled (i.e., fall through spring), we estimate one take of each of these species will occur in this mission area.

Table 7. Estimated Level B Harassment Authorized for Mission Area 7, South China Sea

South China Sea—Mission Area 7 **1 Mission**	
Species	**Take Estimates Level B harassment**
Fin whale	4
Humpback whale	6
North Pacific right whale	1
Western North Pacific gray whale	1
Sperm whale	11

Table 8. Estimated Level B Harassment Authorized for Mission Area 8, Offshore Japan 25 to 40° North

Offshore Japan 25 to 40° North—Mission Area 8 **1 Mission**	
Species	**Take Estimates Level B harassment**
Blue whale	1
Fin whale	1
Sei whale	7
Humpback whale	2
Sperm whale	26
Hawaiian monk seal	4

Table 9. Estimated Level B Harassment Authorized for Mission Area 9, Offshore Japan 10 to 25° North

Offshore Japan 10 to 25° North—Mission Area 9 **1 Mission**	
Species	**Take Estimates Level B harassment**
Blue whale	1
Fin whale	1

Sei whale	69
Humpback whale	9
Sperm whale	24

Table 10. Estimated Level B Harassment Authorized for Mission Area 10, Hawaii North

Hawaii North—Mission Area 10 **1 Mission**	
Species	**Take Estimates Level B harassment**
Blue whale	1
Fin whale	1
Sei whale	3
Humpback whale	13
False killer whale – MHI Insular DPS	2
Sperm whale	17
Hawaiian monk seal	2

Table 11. Estimated Level B Harassment Authorized for Mission Area 11, Hawaii South

Hawaii South—Mission Area 11 **1 Mission**	
Species	**Take Estimates Level B harassment**
Blue whale	1
Fin whale	1
Sei whale	2
Humpback whale	14
False killer whale – MHI Insular DPS	1
Sperm whale	15
Hawaiian monk seal	2

Table 12. Estimated Level B Harassment Authorized for Mission Area 12, Arabian Sea

Arabian Sea—Mission Area 12 **0.5 Mission**	
Species	**Take Estimates Level B harassment**
Blue whale	1
Fin whale	2
Humpback whale	1
Sperm whale	13

Table 13. Estimated Level B Harassment Authorized for Mission Area 13, Andaman Sea

Andaman Sea—Mission Area 13 1 Mission	
Species	**Take Estimates Level B harassment**
Blue whale	1
Fin whale	1
Sperm whale	12

Table 13. Estimated Level B Harassment Authorized for Mission Area 14, Northwest Australia

Northwest Australia—Mission Area 14 0.5 Mission	
Species	**Take Estimates Level B harassment**
Blue whale	1
Fin whale	13
Humpback whale	13
Sperm whale	6
Hawaiian monk seal	2

Table 14. Estimated Level B Harassment Authorized for Mission Area 15, Northeast of Japan

Northeast of Japan—Mission Area 15 0.5 Mission	
Species	**Take Estimates Level B harassment**
North Pacific right whale	2
Western North Pacific gray whale	1
Sperm whale	72
Steller sea lion – Western DPS	1

2.2.2 Mitigation Requirements

The Holder of the Authorization, and any individuals operating under his authority, must conduct the activity identified in 50 CFR § 218.230 and Condition 3 of the Authorization in a manner that minimizes, to the greatest extent practicable, adverse impacts on marine mammals, their habitats, and the availability of marine mammals for subsistence uses. When conducting operations identified in 50 CFR § 218.230, the following mitigation measures must be implemented, reprinted here from the LOA:

(a) The Holder of this Authorization, and any individuals operating under his authority, must not transmit the SURTASS LFA sonar signal at a frequency greater than 500 Hz.

(b) Through mitigation described under 50 CFR § 218.234 and Condition 9 (Section 2.2.2.1; Mitigation Monitoring) of this Authorization, the Holder of this Authorization and any individuals operating under his authority must ensure, to the greatest extent practicable, that no marine mammal is subjected to a sound pressure level of 180 dB re: 1 µPa (rms) or greater.

(c) **LFA Sonar Mitigation Zone**: Prior to commencing and during SURTASS LFA sonar transmissions, the Holder of this Authorization will use near real-time environmental data and underwater acoustic prediction models to determine the propagation of the SURTASS LFA sonar signals in the mission area. The Holder must determine the distance from the SURTASS LFA sonar source to the 180-dB re: 1 µPa (rms) isopleth (*i.e.*, the LFA sonar mitigation zone) to comply with Condition 8(b) (Section 2.2.1.b).

  (i) The Holder of this Authorization will update these sound field estimates every 12 hours or more frequently depending upon changing meteorological or oceanographic conditions.

(d) **Additional One-km Buffer Zone**: The Holder of this Authorization will establish a one-km buffer zone around the LFA sonar mitigation zone.

(e) **Ramp-Up Procedures for the HF/M3 System**: The Holder of this Authorization and any individuals operating under his authority, will ramp up the High Frequency / Marine Mammal Monitoring (HF/M3) active sonar referenced in 50 CFR § 218.234 from a power level beginning at a maximum source sound pressure level of 180 dB re: 1 µPa at 1 m (rms) in ten-dB increments to operating levels over a period of no less than five minutes:

  (i) At least 30 minutes prior to any SURTASS LFA sonar transmission;

  (ii) Prior to any SURTASS LFA sonar calibrations or testing that are not part of regular SURTASS LFA sonar transmissions described in 50 CFR § 218.230; and

  (iii) Anytime after individuals have powered down the HF/M3 active sonar source for more than two minutes.

  Once HF/M3 operators detect a marine mammal, they will not increase the HF/M3 active sonar system's sound pressure level. Resumption of the ramp-up of HF/M3 sonar system would not occur until marine mammals are no longer detected by the HF/M3 active sonar system, passive acoustic monitoring, or visual monitoring described in Condition 9 (Section 2.2.2.1; Mitigation Monitoring).

(f) **Suspension/Delay for SURTASS LFA Sonar Transmissions**: If the Holder of this Authorization, and any individuals operating under his authority, detects a marine mammal through monitoring required under 50 CFR § 218.235 and Condition 9 (Section

2.2.2.1; Mitigation Monitoring) within either the LFA sonar mitigation zone or the 1-km buffer zone, the Holder will immediately suspend or delay SURTASS LFA sonar transmissions.

(g) **Resumption of SURTASS LFA Sonar Transmissions**: The Holder of this Authorization and any individuals operating under his authority may resume/commence SURTASS LFA sonar transmissions 15 minutes after:

(i) All marine mammals have left the area of the LFA sonar mitigation zone and the 1-km buffer zone; and/or

(ii) There is no further detection of any marine mammal within the LFA sonar mitigation zone plus the one-km buffer zone as determined by the passive or active acoustic or visual monitoring protocols described in 50 CFR § 218.235 and Condition 9 (Section 2.2.2.1; Mitigation Monitoring).

(h) **Geographic Restrictions**: The Holder of this Authorization and any individuals operating under his authority will not operate SURTASS LFA sonar such that the SURTASS LFA sonar sound field exceeds 180 dB re: 1 µPa (rms):

(i) At a distance of less than or equal to 22 km (14 miles [mi]; 12 nautical miles [nmi]) from any coastline, including offshore islands.

(ii) At a distance of less than one km (0.62 mi; 0.54 nmi) seaward of the outer perimeter of any Offshore Biologically Important Area (OBIA) for marine mammals designated in 50 CFR § 218.234(f)(2) and described in Condition 8(h)(iii) during the period specified.

(iii) The OBIAs for marine mammals (with specified periods of effectiveness) for SURTASS LFA sonar routine training, testing, and military operations are listed in Table 15. below.

(i) **Operational Exception for SURTASS LFA Sound Field in OBIAs**: During military operations, SURTASS LFA sonar transmissions may exceed 180 dB re: 1 µPa (rms) within the boundaries of an OBIA, including operating within an OBIA, when the Holder of this Authorization determines that it is: 1) operationally necessary to continue tracking an existing underwater contact; or 2) operationally necessary to detect a new underwater contact within the OBIA. This exception does not apply to routine training and testing with the SURTASS LFA sonar systems.

(j) **Mission Planning**: The Holder of this Authorization must maintain a running calculation/estimation of takes of each species and stocks over the effective period of these regulations. The Holder of this Authorization will plan all SURTASS LFA sonar missions to ensure that no more than 12 percent of any marine mammal stock listed in 50 CFR § 218.230(b)(1) through (3) would be taken by Level B harassment annually. This annual per-stock cap of 12 percent applies regardless of the number of SURTASS LFA sonar vessels operating. The Holder of this Authorization must coordinate with the Holder of the Letters of Authorization issued to the USNS *ABLE*, USNS *VICTORIOUS*, USNS *EFFECTIVE*, and the USNS *IMPECCABLE*, to ensure that this condition is met for all vessels combined.

2.2.2.1 Mitigation Monitoring

The Holder of this Authorization, and any individuals operating under his authority, must:

(a) Perform the following for visual mitigation monitoring:

  (i) Marine mammal biologists qualified in conducting at-sea marine mammal visual monitoring from surface vessels will train and qualify designated ship personnel as lookouts to conduct at-sea visual monitoring.

  (ii) Marine mammal biologists will train the lookouts in the most effective means to ensure quick and effective communication within the ship's command structure to facilitate implementation of protective measures if they observe marine mammals.

  (iii) Conduct visual monitoring from the ship's bridge during daylight hours (30 minutes before sunrise until 30 minutes after sunset) during operations that employ SURTASS LFA sonar in the active mode. Maintain a topside watch with standard binoculars (7x) and with the naked eye.

(b) Perform the following for passive acoustic mitigation monitoring:

  (i) Use the low frequency, passive SURTASS sonar system to listen for vocalizing marine mammals.

(c) Perform the following for active acoustic mitigation monitoring:

  (i) Use the HF/M3 active sonar to locate and track marine mammals in relation to the SURTASS LFA sonar vessel and the sound field produced by the SURTASS LFA sonar source array, subject to the ramp-up requirements in § 218.234(e) and Condition 8(e) (Section 2.2.2.e).

Mitigation monitoring under Conditions 9(a), (b), and (c) must:

(a) Commence at least 30 minutes before the first SURTASS LFA sonar transmission (30 minutes before sunrise for visual monitoring);

(b) Continue between sonar transmissions (pings); and

(c) Continue either at least 15 minutes after completion of SURTASS LFA sonar transmission operations (30 minutes after sunset for visual monitoring) or if marine mammals are showing abnormal behavioral patterns, for a period of time until behavior patterns return to normal or conditions prevent continued observations.

Table 15. Offshore Biologically Important Areas.

OBI A No.	Area Name	Water Body	Significant Marine Mammal Species	Seasonal Restrictions
1	Georges Bank	Northwest Atlantic Ocean	North Atlantic right whale	Year-round
2	Roseway Basin Right Whale Conservation Area	Northwest Atlantic Ocean	North Atlantic right whale	June through December
3	Great South Channel, U.S. Gulf of Maine, and Stellwagen Bank National Marine Sanctuary[1]	Northwest Atlantic Ocean/ Gulf of Maine	North Atlantic right whale	January 1 to November 14
4	Southeastern U.S. Right Whale Seasonal Habitat	Northwest Atlantic Ocean	North Atlantic right whale	November 15 to April 15
5	North Pacific Right Whale Critical Habitat	Northeastern Pacific Ocean/Gulf of Alaska and Bering Sea	North Pacific right whale	March through August
6	Silver Bank and Navidad Bank	Northwestern Atlantic Ocean/Caribbean Sea	Humpback whale	December through April
7	Coastal Waters of Gabon, Congo and Equatorial Guinea	Southeastern Atlantic Ocean	Humpback and blue whales	June through October
8	Patagonian Shelf Break	Southwestern Atlantic Ocean	Southern elephant seal	Year-round
9	Southern Right Whale Seasonal Habitat	Southwestern Atlantic Ocean	Southern right whale	May through December
10	Central California National Marine Sanctuaries	Northeastern Pacific Ocean	Blue and humpback whales	June through November
11	Antarctic Convergence Zone	Southern Ocean	Blue, fin, sei, minke, humpback, and Southern right whales	October through March
12	Piltun and Chayvo Offshore Feeding Grounds—Sea of Okhotsk	Northwestern Pacific Ocean/Sea of Okhotsk	Western Pacific gray whale	June through November
13	Coastal Waters off Madagascar	Western Indian Ocean	Humpback and blue whales	July through September for humpback whale breeding / November through December for migrating blue whales

OBIA No.	Area Name	Water Body	Significant Marine Mammal Species	Seasonal Restrictions
14	Madagascar Plateau, Madagascar Ridge, and Walters Shoal	Western Indian Ocean	Pygmy blue, humpback, and Bryde's whales	November through December
15	Ligurian-Corsican-Provençal Basin and Western Pelagos Sanctuary	North-central Mediterranean Sea	Fin whale	July to August
16	Hawaiian Islands Humpback Whale National Marine Sanctuary —Penguin Bank	North-Central Pacific Ocean	Humpback whale	November through April
17	Costa Rica Dome	Eastern Tropical Pacific Ocean	Blue and humpback whales	Year-round
18	Great Barrier Reef Between 16° South and 21° South	Coral Sea/Southwestern Pacific Ocean	Humpback and dwarf minke whales	May through September
19	Bonney Upwelling on the southern coast of Australia	Eastern Indian Ocean	Blue, pygmy blue, and Southern right whales	December through May
20	Northern Bay of Bengal and Head of Swatch-of-No-Ground	Bay of Bengal/Northern Indian Ocean	Bryde's whale (small form)	Year-round
21	Olympic Coast National Marine Sanctuary and the Prairie, Barkley Canyon, and Nitnat Canyon	Northeastern Pacific Ocean	Humpback whale	Olympic Coast National Marine Sanctuary OBIA: December, January, March, and May / The Prairie, Barkley Canyon, and Nitnat Canyon: June to September
22	Abrolhos Bank	Southwest Atlantic Ocean	Humpback whale	August through November

[1] *The boundary of OBIA #3 encompasses the northern critical habitats of the North Atlantic right whale, Stellwagen Bank National Marine Sanctuary, and areas within the Gulf of Maine.*

2.2.3 Monitoring Requirements

Section 101(a)(5)(A) of the MMPA states that in order to issue an LOA for an activity, NMFS must set forth "requirements pertaining to the monitoring and reporting of such taking." The MMPA implementing regulations at 50 CFR § 216.104 (a)(13) indicate that requests for LOAs must include the suggested means of accomplishing the necessary monitoring and reporting that will result in increased knowledge of the species, the level of taking, or impacts on populations of marine mammals that are expected to be present.

The Holder of this Authorization and any individuals operating under his authority for activities described in 50 CFR § 218.230 must:

(a) Cooperate with NMFS and any other federal agency for monitoring the impacts of the activity on marine mammals; and

(b) Designate qualified on-site individuals to conduct the mitigation, monitoring, and reporting activities specified in this Letter of Authorization.

The Holder of this Authorization and any individuals operating under his authority will conduct all monitoring required under the Letter of Authorization to increase knowledge of the affected marine mammal species. The Holder of this Authorization must:

(a) Consider recommendations on the different types of monitoring/research that could increase the understanding of the potential effects of SURTASS LFA sonar transmissions on beaked whales and/or harbor porpoises.

(b) Continue to assess data from the U.S. Navy Marine Mammal Monitoring (M3) program and work toward making some portion of that data, after appropriate security reviews, available to scientists with appropriate clearances. Any portions of the analyses conducted by these scientists based on these data that are determined to be unclassified after appropriate security reviews should be made publicly available.

(c) Continue to explore the feasibility of coordinating with other U.S. Navy fleet assets and/or range monitoring programs to include the use of SURTASS passive sonar (towed horizontal line array) to augment the collection of marine mammal vocalizations before, during, and after designated exercises.

(d) Continue to collect ambient noise data and explore the feasibility of declassifying and archiving the ambient noise data for incorporation into appropriate ocean noise research efforts.

2.2.4 Reporting Requirements

The Holder of this Authorization and any individuals operating under his authority must:

(a) Draft a plan of action outlining a strategy for implementing recommendations on beaked whales and/or harbor porpoise research; or describe in writing why such research is not feasible/or is unlikely to increase the understanding of the potential effects of SURTASS LFA sonar transmissions on beaked whales and/or harbor porpoises, to be followed by a meeting with NMFS to discuss any other potential options.

(b) Provide a status update to NMFS when the Holder of this Authorization submits the next annual application for Authorizations on efforts to assess the data collected by the Marine Mammal Monitoring (M3) program and progress toward making some portion of that data, after appropriate security reviews, available to scientists with appropriate clearances.

(c) Systematically observe SURTASS LFA sonar operations for injured or disabled marine mammals and monitor the principal marine mammal stranding networks and other media to correlate analysis of any whale strandings that could potentially be associated with SURTASS LFA sonar operations. The Holder of this Authorization and any individuals operating under his authority shall:

  (i) Ensure that NMFS is notified immediately, or as soon as clearance procedures allow, if an injured, stranded, or dead marine mammal is observed during or shortly after and in the vicinity of any SURTASS LFA sonar operations. The Holder of this Authorization will report the incident to the Chief, Permits and Conservation Division, Office of Protected Resources, NMFS, at 301-427-8401 and/or by email to Jolie.Harrison@noaa.gov and Dale.Youngkin@noaa.gov.

  (ii) 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).

  (iii) In the event that an injured, stranded, or dead marine mammal is observed by the Holder and any individuals operating under his authority not in conjunction with SURTASS LFA sonar operations, the Holder of this Authorization and any individuals operating under his authority, will report the same information to NMFS as listed above as soon as operationally feasible and clearance procedures allow.

(d) In the event of a ship strike by the SURTASS LFA sonar vessel, at any time or place, the Holder and any individuals operating under his authority, must:

  (i) Immediately, or as soon as clearance procedures allow, report to NMFS the species identification (if known), the size and length of the animal, location (latitude/longitude) of the animal (or the strike if the animal has disappeared), whether the animal is alive or dead (or unknown), including an estimate of its injury status if alive (injured but alive, injured and moving, unknown, etc.).

  (ii) Report the incident to the Chief, Permits and Conservation Division, Office of Protected Resources, NMFS, at 301-427-8401 and/or by email to Jolie.Harrison@noaa.gov and Dale.Youngkin@noaa.gov.

  (iii) Report as soon as feasible to the NMFS the vessel’s name, class/type, and length, as well as operational status, speed and vessel heading.

  (iv) Provide NMFS a photo or video of the struck animal, if equipment is available.

(e) Submit classified and unclassified quarterly mission reports to the Director, Office of Protected Resources, NMFS no later than 30 days after the end of each quarter, beginning on the date of effectiveness of a Letter of Authorization. Each quarterly mission report will include summaries of all active-mode sonar missions completed during that quarter. At a minimum, each classified mission report must contain the following information:

  (i) Dates, times, and location of each vessel during each mission.

  (ii) Information on sonar transmissions during each mission and records of any delays or suspensions.

  (iii) Location of the SURTASS LFA sonar mitigation and buffer zones in relation to the LFA sonar array.

  (iv) Marine mammal observations including animal type and/or species, number of animals sighted, date and time of observations, type of detection (visual, passive acoustic, HF/M3 sonar), bearing and range from vessel, abnormal behavior (if any), and remarks/narrative (as necessary).

  (v) The report will include the U.S. Navy's estimates of the percentages of marine mammal stocks affected (both for the quarter and cumulatively for the year covered by the Authorization) by SURTASS LFA sonar operations (both within and outside the LFA sonar mitigation and buffer zones), using predictive modeling based on mission locations, dates/times of operations, system characteristics, LFA sonar transmission durations, oceanographic environmental conditions, and animal demographics.

  (vi) If no SURTASS LFA sonar missions are completed during a quarter, a report of negative activity will be provided.

(f) Submit an annual, unclassified report to the Director, Office of Protected Resources, NMFS, no later than 45 days after expiration of this Authorization. At a minimum, the annual report will contain the following:

  (i) An unclassified summary of the year's quarterly reports.

  (ii) The U.S. Navy's estimates of the percentages of marine mammal stocks affected by SURTASS LFA sonar operations (both within and outside the LFA sonar mitigation and buffer zones), using predictive modeling based on mission locations, dates/times of operations, system characteristics, LFA sonar transmission durations, oceanographic environmental conditions, and animal demographics.

  (iii) An analysis of the effectiveness of the mitigation measures with recommendations for improvements, where applicable.

  (iv) An assessment of any long-term effects from SURTASS LFA sonar operations.

  (v) Any discernible or estimated cumulative impacts from SURTASS LFA sonar operations.

2.3 Action Area

Action area refers to all areas to be affected directly or indirectly by the proposed Federal action and not merely the immediate area involved in the action (50 CFR § 402.02). The action area for this opinion and conference report includes areas within the Pacific and Indian Oceans:

- East of Japan; north Philippine Sea; west Philippine Sea; offshore Guam; Sea of Japan; East China Sea; South China Sea; offshore Japan (25 to 40° North and 10 to 25° North), and northeast of Japan.
- North and south of the Main Hawaii Islands.
- Arabian Sea, Andaman Sea, and northwest of Australia.

The proposed action may occur in any marine waters within these areas, except the LFA sonar sound field cannot exceed 180 dB re: 1 µPa (rms): 1) at a distance less than 22 km (12 nmi) from any coastline and (2) less than one km (0.5 nmi) from the perimeter of the Hawaiian Islands Humpback Whale National Marine Sanctuary—Penguin Bank OBIA (OBIA #16) located in the north-central Pacific Ocean (Table 15.) from November to April, unless the 145 dB operational exemption applies.

3 APPROACH TO THE ASSESSMENT

Section 7 (a)(2) of the ESA requires Federal agencies, in consultation with NMFS, to insure that their actions either are not likely to jeopardize the continued existence of endangered or threatened species; or adversely modify or destroy their designated critical habitat. The jeopardy analysis considers both survival and recovery of the species. The adverse modification analysis considers the impacts on the conservation value of designated critical habitat.

"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).

3.1 Overview of NMFS' Assessment Framework

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

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) When information is available to do so, we identify the number, age (or life stage), and sex of ESA-listed individuals that are likely to be exposed to the stressors and the populations or subpopulations to which those individuals belong. If it is not practicable to establish the precise number of individuals of the ESA-listed species, we may use a “surrogate” to establish the impact of take on the species (80 FR 26832). 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 to the individuals that have been exposed, the populations those individuals represent, and the species those populations comprise. This is our risk analysis.

7) The adverse modification analysis considers the impacts of the proposed action on the critical habitats essential 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 (RPA) to the action. The RPA must not be likely to jeopardize the continued existence of ESA-listed species nor adversely modify their designated critical habitat and it must meet other regulatory requirements.

3.2 Risk Analysis for Endangered and Threatened Species

Our jeopardy determinations must be based on an action's effects on the continued existence of threatened or endangered species as those "species" have been ESA-listed, which can include true biological species, subspecies, or distinct population segments of vertebrate species. Because the continued existence of ESA-listed species depends on the fate of the populations that comprise them, the viability (that is, the probability of extinction or probability of persistence) of ESA-listed species depends on the viability of the populations that comprise the species. Similarly, the continued existence of populations are determined by the fate of the individuals that comprise them; populations grow or decline as the individuals that comprise the population live, die, grow, mature, migrate, and reproduce (or fail to do so).

Our risk analyses reflect these relationships between listed species and the populations that comprise them and the individuals that comprise those populations. Our risk analyses begin by identifying the probable risks actions pose to listed individuals that are likely to be exposed to an action's effects. Our analyses then integrate those individuals risks to identify consequences to the populations those individuals represent. Our analyses conclude by determining the consequences of those population-level risks to the species those populations comprise.

We measure risks to listed individuals using the individual's "fitness," which are changes in an individual's growth, survival, annual reproductive success, or lifetime reproductive success. In particular, we examine the scientific and commercial data available to determine if an individual's probable response to an Action's effects on the environment (which we identify in our *response analyses*) are likely to have consequences for the individual's fitness.

When individual ESA-listed plants or animals are expected to experience reductions in fitness, those reductions could also reduce the abundance, reproduction rates, or growth rates (or increase variance in one or more of these rates) of the populations those individuals represent (Stearns 1992). Reductions in one or more of these variables (or one of the variables we derive from them) is a *necessary* condition for reductions in a population's viability, which is itself a *necessary* condition for reductions in a species' viability. Therefore, when ESA-listed plants or animals exposed to an Action's effects are *not* expected to experience reductions in fitness, we would not expect that Action to have adverse consequences on the viability of the populations those individuals represent or the species those populations comprise (Anderson 2000; Mills and Beatty 1979; Stearns 1992). As a result, if we conclude that ESA-listed plants or animals are *not* likely to experience reductions in their fitness, we would conclude our assessment because an Action that is not likely to affect the fitness of individuals is not likely to jeopardize the continued existence of ESA-listed species.

If, however, we conclude that individual ESA-listed plants or animals are likely to experience reductions in their fitness, our assessment tries to determine if those fitness reductions are likely to be sufficient to reduce the viability of the populations those individuals represent (measured using changes in the populations' abundance, reproduction, spatial structure and connectivity, growth rates, or variance in these measures to make inferences about the population's extinction risks). In this step of our analyses, we use the population's base condition (established in the *Environmental Baseline* and *Status of Listed Resources* sections of this opinion and conference report) as our point of reference. Finally, our assessment tries to determine if changes in population viability are likely to be sufficient to reduce the viability of the species those populations comprise. In this step of our analyses, we use the species' status (established in the *Status of the Species* section of this opinion and conference report) as our point of reference, and we use our understanding of the general patterns and processes by which species become extinct to help inform our decision about whether changes in the performance of one or more populations are likely to affect the viability of the species those populations comprise.

3.3 Risk Analysis for Designated Critical Habitat

Our "destruction or adverse modification" determinations must be based on an action's effects on the conservation value of habitat that has been designated as critical to threatened or endangered species[3]. If an area encompassed in a critical habitat designation is likely to be exposed to the *direct or indirect consequences of the action on the natural environment*, we ask if essential features included in the designation (if there are any), or physical or biotic phenomena that give the designated area value for conservation, are likely to respond to that exposure.

In our risk analysis for designated critical habitat, we identify (a) the spatial distribution of stressors produced by an action; (b) the temporal distribution of stressors and subsidies produced by an action; (c) changes in the spatial distribution of the stressors with time; (d) the intensity of stressors in space and time; (e) the spatial distribution of physical and biological features of designated critical habitat; and (f) the temporal distribution of constituent elements of designated critical habitat.

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[3] We are aware that several courts have ruled that the definition of destruction or adverse modification that appears in the section 7 regulations at 50 CFR §402.02 is invalid and do not rely on that definition for the determinations we make in this Opinion and conference report. Instead, as we explain in the text, we use the "conservation value" of critical habitat for our determinations which focuses on the designated area's ability to contribute to the conservation or the species for which the area was designated.

If essential features of designated critical habitat (or physical, chemical, or biotic phenomena that give the designated area value for the conservation of ESA-listed species) are likely to respond given exposure to the *direct or indirect consequences of the proposed action on the natural environment*, we ask if those responses are likely to be sufficient to reduce the quantity, quality, or availability of those essential features or physical, chemical, or biotic phenomena.

In this step of our assessment, we must identify or make assumptions about (a) the habitat's probable condition before any exposure as our point of reference (that is part of the impact of the *Environmental Baseline* on the conservation value of the designated critical habitat); (b) the ecology of the habitat at the time of exposure; (c) where the exposure is likely to occur; and (d) when the exposure is likely to occur; (e) the intensity of exposure; (f) the duration of exposure; and (g) the frequency of exposure.

In this step of our assessment, we recognize that the conservation value of designated critical habitat, like the base condition of individuals and populations, is a dynamic property that changes over time in response to changes in land use patterns, climate (at several spatial and temporal scales), ecological processes, and other changes in the biotic components of the habitat, etc. For these reasons, some areas of designated critical habitat might respond to an exposure when others do not. We also consider how designated critical habitat is likely to respond to any interactions and synergisms between or cumulative effects of pre-existing stressors and proposed stressors.

If the quantity, quality, or availability of the essential features of the area of designated critical habitat (or physical, chemical, or biotic phenomena) are reduced, we ask if those reductions are likely to be sufficient to reduce the conservation value of the designated critical habitat for listed species in the action area. In this step of our assessment, we combine information about the contribution of essential features of critical habitat (or of the physical, chemical, or biotic phenomena that give the designated area value for the conservation of ESA-listed species, particularly for older critical habitat designations that have no constituent elements) to the conservation value of those areas of critical habitat that occur in the action area, given the physical, chemical, biotic, and ecological processes that produce and maintain those essential features in the action area. We use the conservation value of those areas of designated critical habitat that occur in the action area as our point of reference for this comparison. For example, if the designated critical habitat in the action area has limited current value or potential value for the conservation of ESA-listed species, that limited value is our point of reference for our assessment.

If the conservation value of designated critical habitat in an action area is reduced, the final step of our analyses asks if those reductions are likely to be sufficient to reduce the conservation value of the entire critical habitat designation. In this step of our assessment, we combine information about the essential features of critical habitat (or of the physical, chemical, or biotic phenomena that give the designated area value for the conservation of ESA-listed species, particularly for older critical habitat designations that have no constituent elements) that are likely to experience changes in quantity, quality, and availability given exposure to an action with information on the physical, chemical, biotic, and ecological processes that produce and maintain those essential features in the action area. We use the conservation value of the entire designated critical habitat as our point of reference for this comparison. For example, if the entire designated critical habitat has limited current value or potential value for the conservation of ESA-listed species, then that limited value is our point of reference for our assessment.

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 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 U.S. Navy SURTASS LFA sonar routine training, testing, and military operations and included this information in the incidental take statement. At such time a 4(d) rule is issued for these species, NMFS and the U.S. Navy may need to reinitiate consultation to amend the incidental take statement.

3.5 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 aspects of the environment. 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), 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 ESA-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 (equals 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.

3.6 Treatment of “Cumulative Impacts” (in the sense of NEPA)

The CEQ 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 ESA-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 ESA-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, testing, and military operations do not exceed maximum levels assessed in the action.

In considering accumulating stressors, 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, transient events), while, conversely, concentrations of toxic chemicals, sediment, and other pollutants accumulate in the environment.

Assessing the response effects of individuals, populations, or species to the accumulation of stressors, 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, forage, migrate, or conduct other biologically important activities); reproductive success; longevity; energy debt, including allostatic loading (wear and tear on the body from repeated chronic stress); body burdens of toxic chemicals; the fitness costs of behavioral decisions; 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 immigrate or emigrate from a population per unit time; mortality within a particular age or stage over generation time; and the reservoir of juveniles in a population that have a high probability of surviving to the age of reproduction (population momentum or its absence).

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

Cumulative effects also include effects of future State, tribal, local, or private actions that are reasonably certain to occur in the action area considered in this opinion and conference report. 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.

3.7 A Brief Background on Sound

Sound is a wave of pressure variations propagating through a medium (for the sonar considered in this opinion, the medium is marine water). Pressure variations are created by compressing and relaxing the medium. Sound measurements can be expressed in two forms: *intensity* and *pressure*. Acoustic intensity is the average rate of energy transmitted through a unit area in a specified direction and is expressed in watts per square meter (W/m^2). Acoustic intensity is rarely measured directly and is derived from ratios of pressures; the standard reference pressure for underwater sound is 1 μPa; for airborne sound, the standard reference pressure is 20 μPa (Richardson et al. 1995d).

Acousticians have adopted a logarithmic scale for sound intensities, which is denoted in decibels (dB). Decibel measurements represent the ratio between a measured pressure value and a reference pressure value (i.e., 1 μPa for underwater sound or 20 μPa for airborne sound). The logarithmic nature of the scale means that each 10 dB increase is a ten-fold increase in power (e.g., 20 dB is a 100-fold increase, 30 dB is a 1,000-fold increase). Humans perceive a ten dB increase in noise as a doubling of sound level or a 10 dB decrease in noise as a halving of sound level. The term “sound pressure level” implies a decibel measure and a reference pressure that is used as the denominator of the ratio (dB re: 1 μPa). Throughout this opinion and conference report, we use 1 micro Pascal (denoted re: 1 μPa) as a standard reference pressure unless noted otherwise.

It is important to note that decibels underwater and decibels in air are not the same and cannot be directly compared. Due to the different densities of air and water and the different reference pressure standards of sound in water and air, a sound with the same intensity (i.e., power) in air and in water would be approximately 63 dB quieter in air. Thus, a sound that is 160 dB loud underwater would have the same effective intensity as a sound that is 97 dB loud in air.

Sound frequency is measured in cycles per second, or Hertz (abbreviated Hz), and is analogous to musical pitch; high-frequency sounds are perceived as high-pitched while low-frequency sounds are perceived as low-pitched sounds. Natural sounds in the ocean span a wide range of frequencies: from earthquake noise at five Hz to harbor porpoise (*Phocoena phocoena*) clicks at 150,000 Hz. These sounds are so low or so high in pitch that humans cannot even hear them; acousticians call these infrasonic and ultrasonic sounds, respectively. A single sound may be made up of many different frequencies. Sounds made up of only a small range of frequencies are called “narrowband” and sounds with a broad range of frequencies are called “broadband;” airguns are an example of a broadband sound source and sonars are an example of a narrowband sound source.

3.8 Evidence Available for the Consultation

In 2002, NMFS’ Endangered Species Division completed its first biological opinion on the U.S. Navy’s proposed employment of the SURTASS LFA sonar system and NMFS’ Permits, Conservation, and Education Division’s proposal to authorize the “take” of marine mammals pursuant to the U.S. Navy's employment of that sonar system. From 2002 to 2015, inclusive, NMFS completed biological opinions (and conference report in 2015) on each annual LOA the Permits and Conservation Division issued to the U.S. Navy for annual SURTASS LFA sonar missions. Additionally, programmatic biological opinions were completed in 2002, 2007, and in 2012. Thus, this opinion and conference report builds upon the earlier biological opinions NMFS has prepared on the employment of the SURTASS LFA sonar system and uses the evidence we collected, analyzed, and synthesized for those earlier opinions as its foundation.

For this current opinion and conference report, we identified new lines of evidence on the potential effects of the SURTASS LFA sonar system on endangered species, threatened species, and critical habitat that has been designated for them. We considered all lines of evidence available through published and unpublished sources that represent evidence of adverse consequences or the absence of such consequences. NMFS' status reviews for ESA-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 *Effects* 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. We also evaluated the quarterly and annual mission reports provided by the U.S. Navy as a requirement of the ESA and MMPA authorizations.

Considering the information that was available, this consultation and our opinion and conference report 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.

3.8.1 The U.S. Navy's Approach to Assessing Behavioral Response of Marine Mammals

This section gives a brief summary on the U.S. Navy's approach to assessing the exposure and behavioral response of marine mammals to SURTASS LFA sonar. The U.S. Navy used the Acoustic Integration Model© (AIM) to simulate the sound field produced by the SURTASS LFA sonar source operations and determine the potential acoustic impacts to marine mammal species present within the sound field of the SURTASS LFA mission areas considered in this opinion and conference report. Further detail on this approach and associated modeling efforts is presented in the 2012 SURTASS LFA FSEIS/SOEIS (Navy 2012a).

The AIM modeling process includes both AIM modeling operations and post-AIM calculations to determine impact risk. During AIM modeling, AIM integrates sound field data generated by an embedded acoustic model are integrated with animal movement data generated from AIM's animal movement engine. The resulting data are stored in files and consist of an exposure history for each simulated animal ("animat"[4]). These data are a sequential history of each acoustic exposure as if each animat was fitted with an "acoustic dosimeter" and the resulting received levels from the LFA sonar source were recorded. These exposure data for individual modeled animats are then scaled and summed to predict the risk of harassment for each animal species.

3.8.1.1 Introduction to AIM

The exact positions of animals relative to sound sources cannot be known. Multiple runs of realistic predictions are therefore used to provide statistical power for the estimated effects. The movement of sources and receivers (animals) are modeled based on measured or defined data. Each source and receiver is modeled via the animat concept. When an animat represents an object, such as an acoustic source, the speed, direction, and depth is usually specified. When an animat represents an animal, movement is defined by specifying behavioral variables, such as dive parameters, swimming speed, and course changes specific to each marine mammal species (see the 2012 SURTASS LFA FSEIS/SOEIS (Navy 2012a) for more details). The results are realistic representations of animal movements such as diving patterns that mimic the real-world diving patterns of that species. The movement of an animat can also be programmed to respond to environmental factors (e.g., water depth at the position of the animat). In this way, marine species that normally inhabit a particular environment can be constrained to stay within a specified marine habitat.

Once the behavior of the animats has been programmed, the simulation is "seeded" with an appropriate number of animats, and the model is run. A model run consists of a user-specified number of steps forward in time. During each time step, each animat is moved according to the programmed rules describing its behavior. For each time step, the received sound level at each receiver animat is calculated. At the end of each time step, each animat evaluates its environment including its three-dimensional (3D) location. If an environmental variable has exceeded the user-specified boundary value (e.g., the animat has moved into water that is too shallow), then the animat will alter its course to react to the environment. These responses to the environment are called "aversions." There are many aversion variables that can be used to specify an animat's reactions and to obtain realistic behavior (e.g., bathymetry, geographic boundaries, water temperature, density of prey species, and level of pollution).

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[4] Animats are computationally simulated animals or objects.

3.8.1.2 Marine Mammal Population Estimation

The distribution of many marine mammal species is irregular and highly dependent upon geography, oceanography, and seasonality. Density and abundance estimates are critical components needed to analytically estimate risk to marine mammal populations from activities occurring in the marine environment. The process for developing density and abundance estimates for every marine mammal species in each mission area was a multi-step procedure that first utilized data with the highest degree of fidelity. Direct estimates from line-transect surveys that occurred in or near each of the 15 mission areas were utilized first. For the majority of species, abundance estimates were available for each of the mission areas. However, density estimates require more sophisticated sampling and analysis and were not always available for each species or stock in all mission areas. When density estimates were not available from a survey in a mission area, then density estimates from a region with similar oceanographic characteristics were extrapolated to the mission area. Further, density estimates are sometimes pooled for species of the same genus if sufficient data are not available to compute a density for individual species or the species are difficult to distinguish at sea.

The marine mammal species potentially occurring at each mission area were modeled in AIM at densities higher than those found in the real world to sufficiently capture the statistical distribution of potential exposure conditions. This “over-population” ensures that the result of the simulation is not unduly influenced by the chance placement of a few animals. To obtain final harassment estimates, the post-processing of the AIM results are normalized by the ratio of the modeled animat density to the real-world animal population density estimate, which allows for greater statistical power without overestimating risk. The resulting harassment estimate is divided by the abundance of the stock to determine the overall percentage of potential risk to the stock of each marine mammal species in each mission area.

3.8.1.3 AIM Modeling for SURTASS LFA Sonar

The simulation areas for acoustic impact analysis were the potentially ensonified areas of the proposed SURTASS LFA mission areas. Each marine mammal species potentially found in these areas was simulated by creating animats programmed with behavioral values describing their dive behavior; including dive depth, surfacing time, dive duration, swimming speed, and course change.

After the animats were created, they were randomly distributed over the simulation area. The simulation area was determined by first finding the range at which the transmission loss was at least 100 dB (more details follow). The time step used for modeling was 30 seconds and the modeling animat density was 0.1 or 0.05 animats/km^2, which is higher than that expected in the actual environment.

During the AIM modeling, the animats were programmed to remain within the analysis area and they were “reflected” off the boundaries of the AIM simulation area. This reflection represented one animat entering the analysis area for each animat leaving the area—hence, a net change in the number of animats in the analysis area was zero and no animats diffused out of the analysis area. For a nominal AIM model run of approximately seven days, it has been the U.S. Navy's experience that only about two to ten percent of the modeled animats encounter the simulation area boundary. Additionally, due to the distance from the simulation area boundary to the source at the box's center (0 to 278 km [0 to 150 nmi]), it is only a very small percentage (typically less than 0.1 percent of all modeled animats) that ever approach within 18.5 (10 nmi) of the source while it is transmitting, within the seven modeled days.

3.8.1.4 Probability of a Behavioral Response

The AIM simulations create a realistic animal movement track for each animat and are based on the best available animal behavioral data. Collectively, the animat tracks derived for each simulation (area/species combination) are representative of the movements of animals in the population under consideration. Within AIM, the acoustic sound field of LFA sonar was also determined and convolved with the animat tracks so that the output of AIM is the time history of exposure for each animat. The cumulative energy was calculated as a 'single ping equivalent' (SPE[5]) and used as input to the risk continuum function to assess the potential risk of MMPA Level B harassment[6].

The U.S. Navy represented the probability of risk (Figure 5), using stressor-response functions generated by mathematical simulation. These functions, which are represented as cumulative probability distributions or cumulative distribution functions, have probability values near zero at very low exposures and probability values near one for very high exposures. From this distribution function, received levels of 150 dB re: 1 µPa at 1 m (rms) generally had a 2.5 percent likelihood of producing a behavioral response that is important to the biology of marine mammals. Received levels of 165 dB re: 1 µ Pa (rms) at 1 m generally had a 50 percent probability of producing such a behavioral response in marine mammals and received levels of 180 dB re: 1 µPa (rms) at 1 m generally have a 95 percent probability of producing such a behavioral response, although this is a generalization for all marine mammals and did not necessarily apply to particular marine mammal species.

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[5] SPE is an intermediate calculation for input to the risk continuum. It accounts for the energy of all LFA sonar transmissions that a modeled animal receives during an entire LFA mission (modeled for operations from 7 to 20 days). SPE is a function of sound pressure level (SPL), not sound exposure level (SEL).

[6] As the statutory definitions are currently applied, a wide range of behavioral reactions may qualify as Level B harassment under the MMPA or harassment pursuant to the ESA, including but not limited to avoidance of the sound source, temporary changes in vocalizations or dive patters, temporary avoidance of an area, or temporary disruption of feeding, migrating, or reproductive behaviors. The U.S. Navy modeled estimates calculated using the risk continuum function do not differentiate between the different types of potential behavioral reactions. For this consultation, we interpret "harass" to mean an intentional or negligent action that has the potential to injure an animal or disrupt its normal behaviors to a point where such behaviors are abandoned or significantly altered.

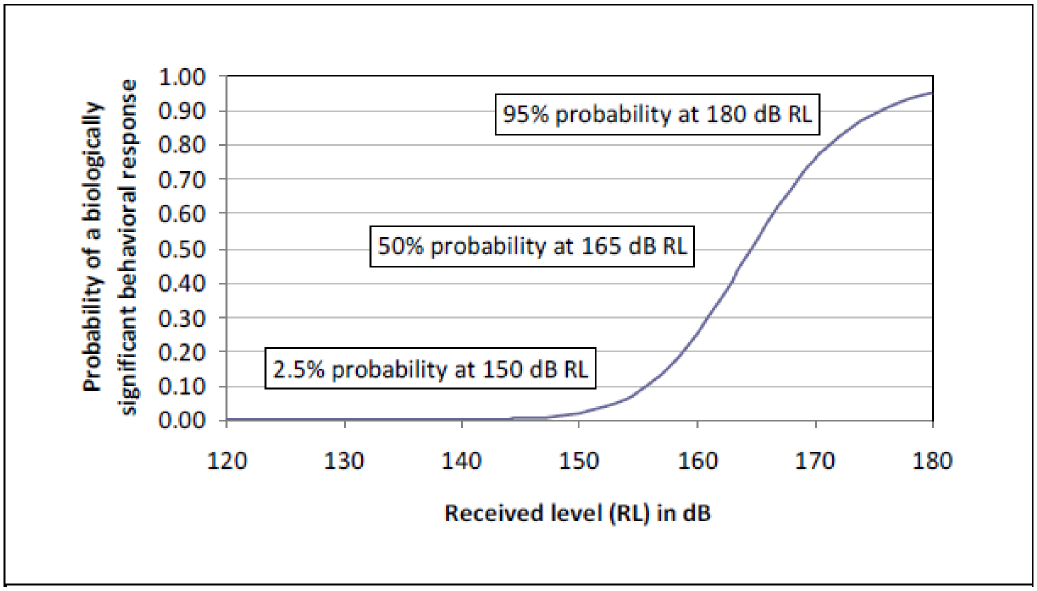

Figure 5. SURTASS LFA Sonar Risk Continuum Function

4 STATUS OF ESA-LISTED RESOURCES

This section identifies the ESA-listed species under NMFS jurisdiction that potentially occur within the action area (as described in Section 2.3) that may be affected by the proposed action (Section 2.1 and 2.2). It then summarizes the biology and ecology of those species and what is known about their life histories in the action area.

For species that are likely to be adversely affected, we describe 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 species' current "reproduction, numbers, or distribution" as described in 50 CFR § 402.02. The section also examines the condition of designated critical habitat throughout the action area, evaluates the conservation value of the various marine environments that make up the action area, and discusses the current function of the essential physical and biological features that provides the conservation value of the designated critical habitat.

Table 16. ESA-listed resources under NMFS' jurisdiction that may occur in the action area.

Species	ESA Status	Critical Habitat	Recovery Plan
Marine Mammals – Cetaceans			
Blue Whale (*Balaenoptera musculus*)	E – 35 FR 18319	N/A	07/1998
Fin Whale (*Balaenoptera physalus*)	E – 35 FR 18319	N/A	75 FR 47538; 08/2010

Species	ESA Status	Critical Habitat	Recovery Plan
Gray Whale (*Eschrichtius robustus*) – Western North Pacific population	E – 35 FR 18319; Eastern North Pacific population was delisted in 1994 – 59 FR 31094	N/A	N/A
Humpback Whale (*Megaptera novaeangliae*)[1]	E – 35 FR 18319	NA	55 FR 29646; 11/1991
- Arabian Sea DPS	PE - 80 FR 22304		
- Western North Pacific DPS	PT - 80 FR 22304		
North Pacific Right Whale (*Eubalaena japonica*)	E – 35 FR 18319 E – 73 FR 12024	71 FR 38277 73 FR 19000	78 FR 34347 06/2013
Sei Whale (*Balaenoptera borealis*)	E – 35 FR 18319	N/A	76 FR 43985 12/2011
Sperm Whale (*Physeter macrocephalus*)	E – 35 FR 18319	N/A	75 FR 81584 12/2010
False Killer Whale (*Pseudorca crassidens*) – Main Hawaiian Islands Insular DPS	E - 77 FR 70915	N/A	N/A
Marine Mammals – Pinnipeds			
Hawaiian Monk Seal (*Neomonachus schauinslandi*)	E - 41 FR 51611	53 FR 18988 80 FR 50925 76 FR 32026[2]	72 FR 46966
Spotted Seal (*Phoca largha*) – Southern DPS	T – 75 FR 65239	N/A	N/A
Steller sea lion (*Eumetopias jubatus*) – Western DPS	E – 55 FR 49204 62 FR 24345	58 FR 45269	3/2008
Fish			
Scalloped Hammerhead Shark (*Sphyrna lewini*) – Indo-West Pacific DPS	T - 78 FR 20718	N/A	N/A
Chum salmon (*Oncorhynchus keta*)		N/A	
- Columbia River ESU	T – 64 FR 14507		2013
- Hood Canal Summer Run ESU	T – 64 FR 14507		2007
Sockeye salmon (*Oncorhynchus nerka*)		N/A	
- Ozette Lake ESU	T – 64 FR 14528		2009
- Snake River ESU	E – 56 FR 58619		2015
Coho salmon (*Oncorhynchus kisutch*)		N/A	
- Lower Columbia River ESU	T - 70 FR 37160		2013
- Oregon Coast ESU	T – 63 FR 42587		N/A
- So. Oregon Northern California Coast ESU	T- 76 FR 50447		2014
- Central California Coast ESU	E – 61 FR 56138		2012

Species	ESA Status	Critical Habitat	Recovery Plan
Steelhead trout (*Oncorhynchus mykiss*) - Lower Columbia River DPS - Upper Columbia River DPS - Middle Columbia River DPS - Puget Sound DPS - Upper Willamette River DPS - Northern California DPS - Central California Coast DPS - Snake River Basin DPS - California Central Valley DPS - South-Central California Coast DPS - Southern California DPS	 T - 71 FR 834 T - 71 FR 834 T - 71 FR 834 T - 72 FR 26722 T - 71 FR 834 T - 71 FR 834 T - 71 FR 834 T - 71 FR 834 T - 71 FR 834 T - 71 FR 834 T - 71 FR 834	N/A	 2013 2007 2009 N/A 2011 N/A N/A N/A 2014 2013 2012
Chinook salmon (*Oncorhynchus tshawytscha*) - California coastal ESU - Central Valley spring-run ESU - Lower Columbia River ESU - Upper Columbia River spring-run ESU - Puget Sound ESU - Sacramento River winter-run ESU - Snake River fall-run ESU - Snake River spring/summer-run ESU - Upper Willamette River spring-run ESU	 T -- 70 FR 37160 T -- 70 FR 37160 T -- 70 FR 37160 T -- 70 FR 37160 T -- 70 FR 37160 T -- 70 FR 37160 T -- 70 FR 37160 T -- 70 FR 37160 T -- 70 FR 37160	N/A	2007 2013 2007 2011 N/A N/A 2014 N/A 2007 2014
Chinese Sturgeon (*Acipenser sinensis*)	E – 79 FR 31222	N/A	N/A
Sakhalin sturgeon (*Acipenser mikadoi*)	E – 79 FR 31222	N/A	N/A
Sea Turtles			
Green Sea Turtle (*Chelonia mydas*) - East Indian-West Pacific DPS - Central North Pacific DPS - Central West Pacific DPS - North Indian DPS	 T – 81 FR 20057 T – 81 FR 20057 E – 81 FR 20057 T – 81 FR 20057	N/A	N/A
Hawksbill Sea Turtle (*Eretmochelys imbricata*)	E – 35 FR 8491	63 FR 46693	63 FR 28359 05/1998
Loggerhead Sea Turtle (*Caretta caretta*) - North Pacific Ocean DPS - North Indian Ocean DPS - Southeast Indo-Pacific Ocean DPS	 E – 76 FR 58868 E – 76 FR 58868 T – 76 FR 58868	N/A	63 FR 28359 05/1998
Olive Ridley Sea Turtle (*Lepidochelys olivacea*) - Breeding Populations on the Pacific Coast of Mexico - All Other Populations	 E - 61 FR 17 T – 43 FR 32800	N/A	63 FR 28359 05/1998

Species	ESA Status	Critical Habitat	Recovery Plan
Leatherback Sea Turtle (*Dermochelys coriacea*)	E – 35 FR 8491 61 FR 17	44 FR 17710 77 FR 4170	63 FR 28359 05/1998

[1] On April 21, 2015, NMFS published its 12-month finding and proposed rule to divide the globally-listed humpback whale into 14 DPSs and list 2 DPSs as endangered and 2 DPSs as threatened under the ESA. The only DPSs proposed for listing that would occur in the action area would be the endangered Arabian Sea DPS and threatened Western North Pacific DPS.
[2] A revision to Hawaiian Monk Seal Critical Habitat was proposed on June 2, 2011.
E = Endangered
T = Threatened
PE = Proposed Endangered
PT = Proposed Threatened
N/A = Not Available
DPS = Distinct Population Segment
ESU = Evolutionarily Significant Unit

4.1 ESA-listed Species Not Likely to be Adversely Affected

NMFS uses two criteria to identify the ESA-listed or designated critical habitat 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. As we do for all ESA consultations, we have applied these criteria in each of the previous programmatic and annual consultations on SURTASS LFA sonar activities. 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 or designated critical habitat. If we conclude that an ESA-listed species or designated critical habitat is not likely to be exposed to the proposed activities, we must also conclude that the species or critical habitat is not likely to be adversely affected by those activities. The second criterion is the probability of a response given exposure. ESA-listed species or designated critical habitat that is exposed to a potential stressor but is likely to be unaffected by the exposure is also not likely to be adversely affected by the proposed action. We applied these criteria to the ESA-listed species in Table 16 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 "taken."

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 an ESA-listed species), but it is very unlikely to occur.

4.1.1 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. 2014a).

4.1.1.1 Populations

On July, 3, 2014 NMFS issued the final determination to list the Central and Southwest (SW) Atlantic DPS and the Indo-West Pacific DPS of scalloped hammerhead shark as threatened species under the 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.

4.1.1.2 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.

4.1.1.3 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 (108 nmi) but this species is also capable of moving much greater distances up to approximately 2,000 km (1,080 nmi). Juvenile movements are likely much shorter distances. Juveniles and adults occur as solitary individuals, pairs, or in schools, and there is evidence of site fidelity to known hot spots.

4.1.1.4 Habitat

Scalloped hammerhead sharks primarily occur in waters over continental and insular shelves and rarely in waters cooler than 22° C. It ranges from surface waters to depths of 512 m (1,680 ft) with occasional dives to deeper water up to 1,000 m (3,281 ft). 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. Scalloped hammerhead sharks appear to prefer waters with stronger currents, greater turbidity, and higher sedimentation and nutrient flow.

4.1.1.5 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 1,000 Hz with the highest sensitivity to sounds at the lower ranges (Casper et al. 2003; Casper and Mann 2006; Myrberg Jr. 2001).

4.1.1.6 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. 2014a) 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. There are no DPS-specific abundance estimates for this species.

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 five 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's 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, Commonwealth of the Northern Mariana Islands (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 a Food and Agriculture Organization (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 two 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.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.

4.1.1.7 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. 2014a) 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.

4.1.1.8 Critical Habitat

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

4.1.1.9 **Conclusion**

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; Olla 1962). As such, this species is anticipated to detect SURTASS LFA sonar (non-impulsive) if exposed. However, the duration and intensity of low-frequency non-impulsive acoustic stressors and the implementation of mitigation (described in Section 6.4.4) will likely minimize the effect this stressor has on scalloped hammerhead sharks.

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/second 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 that included ten pulses/second and 15 to 7.5 decreasing pulses/second.

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.

These signals, some “pulsed,” are substantially different from the SURTASS LFA sonar signals. 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 the fish 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).

Popper et al. (2014) concluded that the relative risk of a fish with no swim bladder illiciting a behavioral reaction in response to low-frequency active sonar was low, regardless of the distance from the sound source. The authors did not find any data on masking by sonar in fishes, but concluded that if it were to occur, masking would result in a narrow range of frequencies being masked (Popper et al. 2014). Popper et al. (2014) also concluded that the risk of mortality, mortal injury, or recoverable injury for fish with no swim bladders exposed to low frequency sonar was low, regardless of the distance from the sound source.

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 SURTASS sonar activities, we do not expect significant responses (including significant behavioral adjustments, TTS, PTS, injury, or mortality). We do not expect injury or mortality to result from any exposures. Additionally, as stated previously, Popper et al. (2014) concluded that behavioral reactions of fish in response to exposure to low-frequency active sonar was unlikely, regardless of the distance from the sound source. The most likely response of scalloped hammerhead sharks exposed to LFA sonar, if any, may be minor temporary changes in their behavior including increased swimming rate, avoidance of the sound source, or changes in orientation to the sound source, none of which rise to the level of take. If these behavioral reactions were to occur, they will not have fitness impacts for the individual, the population, or the DPS.

Therefore, the potential effect of SURTASS LFA sonar on ESA-listed scalloped hammerhead sharks is insignificant. We conclude that the SURTASS LFA sonar training, testing, and military operations the U.S. Navy proposes to conduct in the action area for the period of August 15, 2016, through August 14, 2017, and ongoing for the reasonably foreseeable future is not likely to adversely affect scalloped hammerhead sharks. As a result, scalloped hammerhead sharks will not be considered this further in this opinion and conference report.

4.1.2 Chinese Sturgeon

Chinese sturgeon are large anadromous fish growing up to 449 kilograms (kg) (990 pounds [lb]) and 4.9 m (16 ft) in length. They have gray-black backs, red-brown or gray sides and a white belly. They live to 35 years and are benthic feeders eating primarily insect larvae, fish, and crustaceans.

4.1.2.1 Distribution and Abundance

Chinese sturgeon currently inhabit the Yangtze River and the nearshore areas of the Yellow and East China Seas (Wei 2010). The most recent surveys (2005 to 2007) suggest the rangewide abundance of this species is between 203 to 257 individuals (Wei 2010; Xiao and Duan 2011).

4.1.2.2 Reproduction and Growth

Acipenser sinensis juveniles live in estuaries and near coastlines and migrate upriver when they become sexually mature (Wei 2010). Males reach sexual maturity at eight to 18 years of age and females at 13 to 28 years of age (Wei et al. 1997) Maximum age of reproduction is 35. Adults reach the mouth of the Yangtze River between June and July and reach the middle of the river in

September or October, where they then spawn and overwinter (Wei 2010; Wei et al. 1997). Spawning usually occurs at night in October or November at water temperatures of 15 to 20° C in substrates the size of coarse gravel to 20 to 50 cm (7.9 to 19.7 in) boulders at depths of 8 to 26 m (26 to 85 ft) in current velocities near 1 m/s (Meadows and Coll 2013).

4.1.2.3 Vocalization and Hearing

No information is available on the hearing or vocalization abilities of the Chinese sturgeon. Although Chinese sturgeon have a swim bladder, it is not known to be used in hearing. Popper (2005) reported that studies measuring responses of the ear using physiological methods suggest that a species of the genus *Acipenser* likely is capable of detecting sounds from below 100 Hz to about one kHz, suggesting that sturgeon should be able to localize or determine the direction of origin of sound. Meyer and Popper (2002) recorded auditory evoked potentials of varying frequencies and intensities for lake sturgeon (*Acipenser fulvescens*) and found that lake sturgeon can detect pure tones from 100 Hz to 2 kHz, with best hearing sensitivity from 100 to 400 Hz. Further, the auditory brainstem responses for the lake sturgeon were more similar to goldfish (which is considered a hearing specialist that can hear up to five kHz) (Meyer and Popper 2002). Lovell et al. (2005), using a combination of morphological and physiological techniques, determined that lake sturgeon were responsive to sounds ranging in frequency from 100 to 500 Hz, with the lowest hearing thresholds acquired from frequencies in a bandwidth of between 200 and 300 Hz; lake sturgeon were not sensitive to sound pressure.

Although no information on sound production is available for the Chinese sturgeon, information on vocalizations is known for several other members of the sturgeon family. Lake sturgeon produce low frequency sounds during spawning bouts, principally consisting of drumming sounds that range from five to eight Hz, but low frequency rumbles and hydrodynamic sounds as well as high frequency sounds have also been reported (Bocast et al. 2014). The pallid sturgeon (*Scaphirhynchus albus*) and shovelnose sturgeon (*S. albus*) are known to produce at least four types of sounds during the breeding season, ranging from squeaks and chirps from 1 to 2 kHz, with low frequency knocks and moans ranging in frequency between 90 and 400 Hz (Johnston and Phillips 2003).

4.1.2.4 Status and Trends

Chinese sturgeon were listed as endangered under the ESA in 2014 (79 FR 31222). The population size of *A. sinensis* is decreasing with an estimated 97.5 percent decline in the spawning population over a 37-year period, from approximately 100,000 in the 1970s to approximately 2,200 individuals (95 percent confidence interval of 946 to 4,169) in the early 1980s (Wei 2010). The species was a major commercial fishery resource in the 1960s, but by the end of the 1970s yearly catch had declined to 500 fish (Wei 2010).

Gao et al. (2009) conducted a VORTEX PVA model to estimate the sustainability of the population and to quantify the efficiency of current and proposed conservation procedures. The most likely models predicted the observed decline of Chinese sturgeon resulting from the effect of the Gezhouba Dam and also predicted future declines for the species. The model simulations also demonstrated that the current restocking program is not sufficient to sustain or improve the status of this species, as the capture and handling mortality of the artificial reproduction program induces the loss of more wild mature adults than the recruitment expected by the artificial reproduction. Thus stocking programs intended to help the species can have a net negative effect.

4.1.2.5 Threats

The construction of the Gezhouba Dam limits the distribution of *A. sinensis* in the Yangtze River (Wei 2010; Zenglong 1998) and affects recruitment and reproductive development (Wei et al. 1997). Historically, the spawning habitats of Chinese sturgeon were located in the main stream of the upper Yangtze and the lower Jinsha rivers, covering a stretch of about 800 km (432 nmi) of river length. However, after the damming their spawning areas were limited to a 30 km reach below the Gezhouba Dam (Wei et al. 1997), with only two favorable sites being established below the dam (Ban et al. 2011). The completion of the Three Gorges Dam upstream of the Gezhouba dam in 2003 has further impacted the species by lowering the water level of the Yangtze River in fall and winter and affecting the water temperature and other stream characteristics (Wei 2010; Xiao and Duan 2011). Three Gorges Dam, the world's largest, and only fully operational in 2010, also reduces the average discharge of the Yangtze by 40 percent, and this is expected to seriously affect the remaining spawning habitat into the future. The dams have a serious effect on spawning (Meadows and Coll 2013). A proposed hydroelectric project on the Pearl River, the Changzhou Dam, will block spawning migrations in that system (Wei et al. 1997). Water pollution is also a problem for the species, especially in the Yangtze River, as much untreated wastewater discharges into the river each year (Xue et al. 2008).

Acipenser sinensis was a major commercial fishery resource in the 1960s, but by the end of the 1970s catch had declined to 500 fish and has not recovered (Wei 2010). Drift nets were used to catch it in the river and set nets were used at the river mouth (Wei 2010). Commercial fishing has been prohibited since 1983 (Billard and Lecointre 2001).

Introduced exotic sturgeon in the Yangtze River are an identified threat to *A. sinensis* (Li et al. 2009). Since the end of the 1990s, farmers began cage-farming many exotic sturgeon species in the Yangtze River (Shi et al. 2002; Wei et al. 1997). None of these legally farmed sturgeons (including *A. schrenckii*, *H. dauricus,* and their hybrids) are native to the Yangtze River system, so they could compete with native sturgeon. In 2006 the *A. sinensis* Emergency Center (Changshu City, Jiangsu Province) collected 221 young sturgeon from their fishery resources monitoring nets in the Yangtze River. Seventy percent were hybrids, while only 30 percent were pure *A. sinensis* (Chen 2007).

The long lifespan and late maturation of *A. sinensis* make it susceptible to overexploitation. Zhang et al. (2000) screened the nuclear genomes of 70 samples collected in the Yangtze River from 1995 to 1997 and found low genetic variability. Ship strikes and excessive sound have also been noted as threats for this species (Wang et al. 2011b).

4.1.2.6 Critical Habitat

No critical habitat has been designated for this species.

4.1.2.7 Conclusion

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 did not find any data on masking by sonar in fishes, but concluded that if it were to occur, masking would only occur in a narrow range of frequencies being masked by the sonar transmissions (Popper et al. 2014).

The precise expected response of Chinese sturgeon 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 SURTASS sonar activities, and the concentration of Chinese sturgeon in nearshore coastal or freshwater areas, we do not expect frequent exposure or significant responses from any exposures (including significant behavioral adjustments, TTS, PTS, injury, or mortality). We do not expect injury or mortality to result from any exposures. Additionally, as stated previously, Popper et al. (2014) concluded that behavioral reactions of fish in response to exposure to low-frequency active sonar was unlikely, regardless of the distance from the sound source. The most likely response of Chinese sturgeon exposed to LFA sonar, if any, may be minor temporary changes in their behavior including increased swimming rate, avoidance of the sound source, or changes in orientation to the sound source, none of which rise to the level of take. If these behavioral reactions were to occur, they will not have fitness impacts for the individual, the population, or the DPS.

Therefore, the potential effect of SURTASS LFA sonar on ESA-listed Chinese sturgeon is insignificant. We conclude that the SURTASS LFA sonar training, testing, and military operations the U.S. Navy proposes to conduct in the action area for the period of August 15, 2016, through August 14, 2017, and ongoing for the reasonably foreseeable future is not likely to adversely affect Chinese sturgeon. As a result, Chinese sturgeon are not considered further in this opinion and conference report.

4.1.3 Sakhalin Sturgeon

Sakhalin sturgeon are large anadromous fish attaining weights up to 149.7 kg (330 lb) and lengths up to 2.4 m (eight ft). They have dark green backs, an olive-green stripe on their sides, and a yellowish green-white belly. They are benthic feeders, primarily eating invertebrates and other fish.

4.1.3.1 Distribution and Abundance

Currently, Sakhalin sturgeon are found throughout the Sea of Okhotsk, in the Sea of Japan as far east as the eastern shore of Hokkaido (Japan), along the Asian coast as far south as Wonsan (North Korea), and to the Bering Strait on the coast of the Kamchatka Peninsula (Mugue 2010;

Shmigirlov et al. 2007). It spawns persistently only in the Tumnin River in the Khabarovsk Region in Russia (Shmigirlov et al. 2007), though at least one mature female was caught in Bay Viyakhtu near the settlement of Trambus in the summer of 2010, and a mature male was caught in the Viyakhtu River in 2011 (Koshelev et al. 2012).

The most recent population estimates range from ten to 30 adults entering the Tumnin River to spawn annually, with only three specimens caught in 2005, and two in 2008. These few specimens were used to establish aquaculture stocks (Mugue 2010). Koshelev et al. (2012) report catches of 17 individuals in the Tumnin River and Datta Bay from 2006 to 2008. Recent seine fish surveys in the Tumnin River during the past two years have not caught this species (Zolotukhin 2012). Five to ten Sakhalin sturgeon are caught annually in the Amur River estuary where they were introduced (Krykhtin and Svirskii 1997). The species is now ESA-listed as extinct in the Hokkaido Red Data Book in Japan (Omoto et al. 2004).

4.1.3.2 Reproduction and Growth

Acipenser mikadoi has an estimated generation length of 15 years and reaches maturity between eight to ten years of age. They spawn in June through July in the Tumnin River, and in April and May in the rivers of Hokkaido, Japan (Mugue 2010), with migration occurring once individuals reach 135 cm (53 in) total length Koshelev et al. (2012). Spawning occurs at water temperatures of 7.2 to 11.5° C, and juveniles migrate to the sea in the fall of the same year they hatched (Birstein 1993). Estuaries are thought to be the nursery grounds for the species (Paul 2007).

4.1.3.3 Vocalization and Hearing

No information is available on the hearing or vocalization abilities of the Sakhalin sturgeon but some limited information is available on other sturgeon species. Sakhalin sturgeon have a swim bladder but it is not known to be used for hearing. Popper (2005) reported that studies measuring responses of the ear using physiological methods suggest that a species of the genus *Acipenser* likely is capable of detecting sounds from below 100 Hz to about 1 kHz, suggesting that sturgeon should be able to localize or determine the direction of origin of sound. (Meyer and Popper 2002) recorded auditory evoked potentials of varying frequencies and intensities for lake sturgeon (*Acipenser fulvescens*) and found that lake sturgeon can detect pure tones from 100 Hz to 2 kHz, with best hearing sensitivity from 100 to 400 Hz. Further, the auditory brainstem responses for the lake sturgeon were more similar to goldfish (which is considered a hearing specialist that can hear up to five kHz) (Meyer and Popper 2002). Lovell et al. (2005), using a combination of morphological and physiological techniques, determined that lake sturgeon were responsive to sounds ranging in frequency from 100 to 500 Hz, with the lowest hearing thresholds acquired from frequencies in a bandwidth of between 200 and 300 Hz; lake sturgeon were not sensitive to sound pressure.

Although no information on sound production is available for the Sakhalin sturgeon, other members of the sturgeon family are known to produce sounds, apparently especially during spawning. Lake sturgeon produce low frequency sounds during spawning bouts, principally consisting of drumming sounds that range from 5 to 8 Hz, but low frequency rumbles and hydrodynamic sounds as well as high frequency sounds have also been reported (Bocast et al. 2014). The pallid sturgeon (*Scaphirhynchus albus*) and shovelnose sturgeon (*S. albus*) are known to produce at least four types of sounds during the breeding season, ranging from squeaks and chirps from one to two kHz, with low frequency knocks and moans ranging in frequency between 90 and 400 Hz (Johnston and Phillips 2003).

4.1.3.4 Status and Trends

Sakhalin sturgeon were listed as endangered under the ESA in 2014 (79 FR 31222). The population size of *A. mikadoi* is decreasing and has been declining over the past century (Mugue 2010). Anecdotal reports note that the species "was common in the fish markets of Japan in the 1950s and now only a few specimens are found per year" (Mugue 2010).

4.1.3.5 Threats

Pollution from agriculture, oil production, and mining is degrading habitat quality for *A. mikadoi* (Mugue 2010; Shilin 1995). Logging also occurs along the Tumnin River (Erickson 2005). Damming of the Tumnin River is under discussion; this would massively affect the reproduction of this species.

Acipenser mikadoi was harvested commercially in the past and illegal poaching continues to be a threat (Mugue 2010; Shilin 1995). Bycatch from salmon trawling off the coast is also a threat (Mugue 2010; Shilin 1995).

4.1.3.6 Critical Habitat

No critical habitat has been designated for this species.

4.1.3.7 Conclusion

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 did not find any data on masking by sonar in fishes, but concluded that if it were to occur, masking would only occur during in a narrow range of frequencies being masked by the sonar transmissions (Popper et al. 2014).

The precise expected response of Sakhalin sturgeon 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 SURTASS sonar activities, and the concentration of Sakhalin sturgeon in nearshore coastal or freshwater areas, we do not expect frequent exposure or significant responses from any exposures (including significant behavioral adjustments, TTS, PTS, injury, or mortality). We do not expect injury or mortality to result from any exposures. Additionally, as stated previously, Popper et al. (2014) concluded that behavioral reactions of fish in response to exposure to low-frequency active sonar was unlikely, regardless of the distance from the sound source. The most likely response of Sakhalin sturgeon exposed to LFA sonar, if any, may be minor temporary changes in their behavior including increased swimming rate, avoidance of the sound source, or changes in orientation to the sound source, none of which rise to the level of take. If these behavioral reactions were to occur, they will not have fitness impacts for the individual, the population, or the DPS.

Therefore, the potential effect of SURTASS LFA sonar on ESA-listed Sakhalin sturgeon is insignificant. We conclude that the SURTASS LFA sonar training, testing, and military operations the U.S. Navy proposes to conduct in the action area for the period of August 15, 2016, through August 14, 2017, and ongoing for the reasonably foreseeable future is not likely to adversely affect Sakhalin sturgeon. As a result, Sakhalin sturgeon are not considered further in this opinion and conference report.

4.1.4 ESA-Listed Salmonids

ESA-listed salmonids (i.e., all Chinook, chum, coho, and sockeye ESUs and steelhead DPSs) may occur in the action area during their ocean migrations. Coded-wire tag data indicates that the majority of individual Chinook, coho, sockeye, and chum would not migrate into areas of the western North Pacific that overlap with SURTASS LFA mission areas (Masuda et al. 2015). ESA-listed steelhead travel much farther westward during their ocean migrations and would be more likely to occur in SURTASS LFA mission areas (Myers et al. 2006; Myers et al. 1996). Any ESA-listed salmonids that occur in the action area are likely to occur only in the more northerly SURTASS LFA mission areas (i.e., mission areas 8, 15, and possibly 9) in the western North Pacific. Further, these species likely occur in very low abundances in this farthest portion of their distributional extent.

Hastings et al. (1996) studied the effects of low frequency underwater sound on fish hearing. The authors exposed the teleost fish *Astronotus ocellatus* to sound of varying frequencies (60 or 300 Hz), duty cycles (20 percent or continuous), and intensity (100, 400, or 180 dB re 1uPa). The only treatment where the authors observed some limited damage to sensory hair cells was with one hour of continuous exposure at 300 Hz and 180 dB, but this result was inconsistent. The authors recommended caution if attempting to extrapolate this result to other species or other sound sources, and also suggested that damage would be even more limited with shorter term stimulation or if fish were free to leave the site of stimulation. For the 2015 consultation on the the Navy’s Northwest Training and Testing activities (NMFS 2015), NMFS and the Navy developed sound exposure criteria for low frequency sonar for fish. Though SURTASS LFA sonar was not evaluated, for other low frequency sonar sources (i.e., low-frequency sources equal to 180 dB and up to 200 dB) it was determined that fish would need to be within one meter of the sonar source in order to experience TTS.

Popper et al. (2008; 2007; Popper and Hastings 2009b) investigated the effects of exposing several fish species, including a salmonid, to SURTASS LFA sonar, focusing on the hearing and on non-auditory tissues. Their 2007 study exposed the fish to LFA sonar pulses for time intervals that would be substantially longer than what would occur in nature, but the fish did not experience mortalities or damage to body tissues at the gross or histological level. Some fish experienced temporary losses in hearing sensitivity, but they recovered within several days of exposure. Popper et al. (2014) did not find any data on masking by sonar in fishes, but concluded that if it were to occur, masking would result in a narrow range of frequencies being masked (Popper et al. 2014). Popper et al. (2014) also concluded that the risk of immediate mortality, mortal injury, or recoverable injury for fish with swim bladders not involved in hearing exposed to low frequency sonar was low, regardless of the distance from the sound source. Low frequency sonar lacks the fast rise times, high peak pressures, and high acoustic impulse that could lead to mortality or injury in fish.

ESA-listed fish in relatively close proximity to the LFA sonar source (e.g., within one meter where TTS could occur) would likely move to actively avoid being within one meter of the source because they would likely perceive the sonar source as a potential predator. Additionally, any hearing loss would be temporary, and individuals would be expected to fully recover shortly after exposure (Lombarte et al. 1993; Smith et al. 2006). Also, the ESA-listed salmonid species considered in this opinion lack notable hearing specialization, which minimizes the likelihood of any instance of TTS affecting an individual’s fitness. To our knowledge, no studies have examined the fitness implications when a fish, without notable hearing specialization, experiences TTS. Popper et al. (2014) suggested that fishes experiencing TTS may have a decreased ability to communicate, detect predators or prey, or assess their environment. However, the species considered in this opinion are able to rely on alternative mechanisms (e.g., sight, lateral line system) to detect prey, avoid predators, spawn, and to orient in the water column (Popper et al. 2014). Additionally, hearing is not thought to play a role in salmon migration (e.g., Putnam et al. 2013). Because these species are able to rely on alternative mechanisms for these essential life functions, instances of TTS would not likely disturb a fish to such an extent as to significantly disrupt normal behavioral patterns, including breeding, feeding, or sheltering and would not rise to the level of take. Therefore, the effect of exposure to LFA sonar that may result in TTS is insignificant and is not likely to adversely affect the ESA-listed fish species considered in this opinion.

We also assessed the potential for exposure to non-impulsive acoustic stressors to result in behavioral responses. ESA-listed salmonids would be expected to be able to detect low-frequency sources. 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). However, Popper et al. (2014) concluded that the relative risk of a fish eliciting a behavioral reaction in response to low-frequency sonar was low, regardless of the distance from the sound source. Should an ESA-listed fish elicit a behavioral reaction from exposure to low-frequency sonar, we do not expect these reactions to have any measurable effects on any individual's fitness. We expect individuals that exhibit a temporary behavioral response would return to pre-exposure behavior immediately following each exposure. Exposure time is expected to be brief because both the vessel and the individual fish are moving while in the ocean environment. Similar to instances of TTS, these short term behavioral reactions would not likely disturb a fish to such an extent as to significantly disrupt normal behavioral patterns, and such reactions would not rise to the level of take.

Therefore, the potential effect of SURTASS LFA sonar on ESA-listed salmonid species (i.e., all Chinook, chum, coho, and sockeye ESUs and steelhead DPSs) is insignificant. We conclude that the SURTASS LFA sonar training, testing, and military operations the U.S. Navy proposes to conduct in the action area for the period of August 15, 2016, through August 14, 2017, and ongoing for the reasonably foreseeable future is not likely to adversely affect ESA-listed salmonid species. As a result, ESA-listed salmonids will not be considered further in this opinion and conference report.

4.1.5 North Pacific Right Whale Critical Habitat

In July 2006, NMFS designated two areas as critical habitat for right whales in the North Pacific (71 FR 38277). The areas encompass about 95,182 km^2 (36,750 mi^2) of marine habitat, which include feeding areas within the Gulf of Alaska and the Bering Sea that support the species. The primary conservation value of this critical habitat is the presence of large copepods and oceanographic factors that concentrate this prey of North Pacific right whales. The Bering Sea is part of the polar non-operational area for SURTASS LFA sonar. During 2016 to 2017, SURTASS LFA sonar will not operate in the Gulf of Alaska. We expect any SURTASS LFA sonar signals that may propagate within proximity of North Pacific right whale designated critical habitat (closest mission area is Hawaii North) will have been reduced to negligible sound levels due to transmission loss. Therefore, the potential effect of SURTASS LFA sonar on North Pacific right whale designated critical habitat is insignificant. We conclude that the SURTASS LFA sonar training, testing, and military operations the U.S. Navy proposes to conduct in the action area for the period of August 15, 2016 through August 14, 2017 pursuant to the five-year MMPA rule (August 15, 2012 through August 14, 2017) and ongoing for the reasonably foreseeable future will not destroy or adversely modify the critical habitat that has been designated for North Pacific right whales. As a result, designated critical habitat of North Pacific right whales will not be considered further in this opinion and conference report.

4.1.6 Hawaiian Monk Seal Critical Habitat

Designated critical habitat that has been designated for Hawaiian monk seals occurs in the action area for SURTASS LFA sonar. Critical habitat was originally designated on April 30, 1986 (51 FR 16047) and was extended on May 26, 1988 (53 FR 18988; CFR § 226.201). Designated critical habitat includes all beach areas, sand spits and islets, including all beach crest vegetation to its deepest extent inland, lagoon waters, inner reef waters, and ocean waters out to a depth of 20 fathoms (37 m [121 ft]) around the following land areas: Kure Atoll, Midway Islands, except Sand Island and its harbor, Pearl and Hermes Reef, Lisianski Island, Laysan Island, Maro Reef, Gardner Pinnacles, French Frigate Shoals, Necker Island, and Nihoa Island. The marine component of this habitat was designated primarily as feeding areas for Hawaiian monk seals, while terrestrial habitat serves as pupping and nursing habitat for mothers and pups. Because the marine critical habitat is located within the Papahānaumokuākea Marine National Monument, fishing is forbidden within designated critical habitat.

On 2 June 2011, NMFS proposed to extend critical habitat in the Northwestern Hawaiian Islands (NWHI) to Sand Island (Midway) and ocean waters out to a depth of 500 m (1,640 ft) (76 FR 32026). Additionally, critical habitat was proposed to be extended in the areas around the Main Hawaiian Islands (MHIs): Kaula Island, Niihau, Kauai, Oahu, Maui Nui (including Kahoolawe, Lanai, Maui, and Molokai), and Hawaii (except areas designated for military use as specified in the proposed rule) to a depth of 500 m and inland to a distance of 5 m (16.4 ft) from shore. In the Draft Biological Report for the Revision of Critical Habitat for Hawaiian Monk Seals (NMFS 2010b), the Critical Habitat Review Team identified five features essential to the conservation of the species: 1) areas with characteristics preferred by monk seals for pupping and nursing, 2) shallow, sheltered aquatic areas adjacent to coastal locations preferred by monk seals for pupping and nursing, 3) marine areas from 0 to 500 m in depth preferred by juvenile and adult monk seals for foraging, 4) areas (terrestrial areas for haul out) with low levels of anthropogenic disturbance, and 5) marine areas with adequate prey quantity and quality. Due to the operational restrictions for SURTASS LFA sonar, sound levels above 180 dB will not be generated within 22 km (12 nmi) of any coastline or land area. Terrestrial areas that serve as pupping and nursing habitat for mothers and pups would not be exposed to sonar signals as the SURTASS LFA array is deployed below the water column. Additionally, the operational restrictions on the use of SURTASS LFA sonar would prevent the highest sound levels from LFA sonar signals from penetrating shallow, sheltered aquatic areas with appreciable signal strengths. Therefore, essential features associated with terrestrial and shallow water habitat will not be affected by the proposed action.

Features essential to the conservation of Hawaiian monk seals relevant to routine training, testing, and military operation of SURTASS LFA sonar include marine areas for foraging with adequate prey quality and quantity. Foraging habitat may range from barrier reefs, leeward slopes of reefs and islands, submarine ridges, nearby seamounts, submerged reefs and banks, and deep coral beds. Preferred foraging habitat of adult monk seals is characterized by sand terraces and talus slopes. These habitats provide substrate and materials for preferred benthic and cryptic prey species to hide. Prey resources may include a variety of species including some benthic and offshore teleosts, cephalopods, and crustaceans.

Studies using active acoustics indicate that sound sources could have indirect, adverse effects on marine animals by reducing the abundance or availability of prey or changing the structure or composition of the fish community. Hawaiian monk seals could forage in some portion of the action area. The indirect effects of LFA sonar could occur if fish or invertebrates experience lethal or sub-lethal damage, stress responses, or alterations in their behavior or distribution in response to acoustic energy produced by operation of SURTASS LFA. Because fish and invertebrate species are prey for Hawaiian monk seals, such effects might have adverse consequences for individuals foraging in the action area.

A number of investigators have suggested that fish exposed to high intensity sounds could show a range of non-auditory injuries, extending from the cellular level to gross damage of the swim bladder and circulatory system (Popper and Hastings 2009a). However, the bulk of the data suggesting such injuries come from studies that tested the effects of explosives on fish (Hastings and Carlson 2008; Popper and Hastings 2009a; Popper and Hastings 2009c; Yelverton et al. 1975). Similar results have been observed in fish exposed to the impulsive sounds from pile driving when fish are at an undetermined range but very close to the pile driving source (Abbott and Bing-Sawyer 2002; CalTrans 2004). There is no evidence for such damage when fish are exposed to sounds similar to those produced by low-frequency sonars.

A few studies have examined the effects of SURTASS LFA sonar and seismic airguns on non-auditory tissues of fish (Kane et al. 2010; Popper 2005; Popper et al. 2007). The significant point from these studies is that neither source, despite being very intense, had any effect on non-auditory tissues. In all fish, the swim bladder was intact after exposure, and in the one study that involved an expert fish pathologist (to ensure that the non-auditory tissues of the fish sacrificed were examined properly), there was no damage to tissues either at the gross or cellular levels (Kane et al. 2010; Popper et al. 2007). These studies provide the first direct evidence that exposure to SURTASS LFA sonar is not likely to cause death or damage to fish. However, it must be cautioned, as done by Hastings et al. (1996), McCauley et al. (2003), Popper et al. (2007), and Kastelein et al. (2008) (among others) that all studies to date have been done with a very limited number of species and extrapolation among species and to other sound sources (or even to other levels or durations of the same sound sources), must be done with extreme caution, at least until there are more data upon which to base any extrapolations. Other studies have similarly found no mortality resulting from exposure to high intensity sounds (Hastings and Carlson 2008; Kane et al. 2010; McCauley et al. 2000; McCauley et al. 2003; Popper 2005; Popper et al. 2007). In contrast, one report by Turnpenny et al. (1994) suggested that sound exposure could produce substantial damage in caged fish. However, reviews by subject matter experts found problems with this report and concluded that it did not appear to reflect the best available science on this issue.

In their "*Sound Exposure Guidelines for Sea Turtles and Fishes*," a technical report developed and approved by Accredited Standards Committee S3/SC 1 Animal Bioacoustics, 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. Additionally, the authors did not find any data on masking by sonar in fishes, but concluded that if it were to occur, masking would only occur during in a narrow range of frequencies being masked by the sonar transmissions (Popper et al. 2014).

A range of invertebrates are reported to be sensitive to low-frequency (10 to 150 Hz) hydroacoustic disturbances induced by sound waves or other sources – e.g., jelly fish, crustaceans, arrow worms, octopus, and squid (Western Australian Department of Industry and Resources 2002). This sensitivity overlaps the frequency range of SURTASS LFA sonar (100 to 500 Hz), indicating that some mobile invertebrates could perceive SURTASS LFA sonar activity (Western Australian Department of Industry and Resources 2002). Available studies report responses to airgun shots as being limited to transient alarm responses such as tail-flicks (lobsters) or siphon closing (ascidians) (Western Australian Department of Industry and Resources 2002), although mortality of giant squid in the Bay of Biscay may possibly have been linked to seismic airgun activity in the area (Guerra et al. 2004). McCauley *et al*. (2000b; 2000a) examined the effect of marine seismic surveys on captive squid and cuttlefish and reported a strong startle response or directed movement away from airguns during sudden, nearby start-ups at received levels of 174 dB re: 1 μPa (rms). Alarm responses in squid were detected during gradual ramp-up of airguns once levels exceeded 156 to 161 dB re: 1 μPa (rms). Squid in these trials appeared to make use of the sound shadow measured near the water surface. These responses for captive squid suggest that behavioral changes and avoidance of operating airguns would likely occur. The authors concluded squid significantly alter their behavior at an estimated distance of two to five km (1.1 to 2.7 nmi) from an approaching large seismic source. A more recent study by Andre et al. (2011) exposed four species of squid and octopus in a tank to two hours of intense sounds. The authors reported tissue degeneration cause by the sounds. However, this study had substantial design and control problems and the results could have been attributed to those problems. Based on a review of airgun impacts on invertebrates by Christian and Bocking (2010), studies to date have not revealed any consistent evidence of serious pathological or physiological effects on invertebrates.

These studies indicate that exposure to low frequency sound has limited potential to affect fish and invertebrates. Available evidence does not suggest low-frequency non-impulsive sound sources would be expected to cause mortality or physiological damage. Additionally, as indicated by Popper et al. (2014), the relative risk of a fish illiciting a behavioral reaction in response to low-frequency sonar is low, regardless of the distance from the sound source. Though squid and other invertebrates appear to exhibit alarm responses and avoidance of sound sources, individuals would be expected to resume normal behaviors immediately after initial exposure. We do not expect any such responses to impact the feeding ability of Hawaiian monk seals in the action area.

Based on the above, we consider any potential effects of the proposed action on essential features of Hawaiian Monk Sea Critical Habitat to be discountable (for terrestrial and shallow water areas used for pupping, nursing, and haulout) insignificant (marine areas for foraging with adequate prey quality and quantity). Nearly all the designated critical habitat for the Hawaiian monk seal lies within the coastal standoff distance for SURTASS LFA sonar. A small area of the Hawaiian monk seals designated critical habitat at Penguin Bank extends beyond the coastal standoff distance, but his area is within an OBIA for SURTASS LFA sonar. Thus, the portion of the designated critical habitat that extends beyond the coastal standoff distance is protected as an area wherein SURTASS LFA sonar cannot exceed 180 dB within 1 km (0.5 mi) of the OBIA boundary. Penguin Bank is also part of the Hawaiian Islands Humpback Whale National Marine Sanctuary. We conclude that the SURTASS LFA sonar training, testing, and military operations the U.S. Navy proposes to conduct in the action area for the period of August 15, 2016 through August 14, 2017 pursuant to the five-year MMPA rule (August 15, 2012 through August 14, 2017) and ongoing for the reasonably foreseeable future will not destroy or adversely modify the critical habitat that has been designated for Hawaiian monk seals.

Additionally, if the proposed rulemaking becomes effective extending Hawaiian monk seal critical habitat and if the proposed listings are finalized as proposed and assessed in this opinion and conference report, we conclude that the SURTASS LFA sonar training, testing, and military operations the U.S. Navy proposes to conduct in the action area for the period of August 15, 2016 through August 14, 2017 pursuant to the five-year MMPA rule (August 15, 2012 through August 14, 2017) and ongoing for the reasonably foreseeable future would not destroy or adversely modify the critical habitat extension that has been proposed. As a result, designated critical habitat of Hawaiian monk seals will not be considered further in this opinion and conference report.

4.1.7 Critical Habitat of Steller Sea Lion - Western DPS

Critical habitat was designated on August 27, 1993, for both Eastern and Western DPS Steller sea lions in California, Oregon, and Alaska (58 FR 45269). Steller sea lion designated critical habitat includes all major rookeries in California, Oregon, and Alaska as well as major haulouts (sites that provide regular retreat from the water on exposed rocky shoreline, gravel beaches, and wave-cut platforms or ice) in Alaska and includes a 37 km (20 nmi) buffer around these locations. Essential features of Steller sea lion designated critical habitat include the physical and biological habitat features that support reproduction, foraging, rest, and refuge, and include terrestrial, air and aquatic areas. Specific terrestrial areas include rookeries and haul-outs where breading, pupping, refuge and resting occurs. More than 100 major haulouts are documented. The principal, essential aquatic areas are the nearshore waters around rookeries and haulouts, their forage resources and habitats, and traditional rafting sites. Air zones around terrestrial and aquatic habitats are also designated as critical habitat to reduce disturbance in these essential areas. Specific activities that occur within the habitat that may disrupt the essential life functions that occur there include: (1) wildlife viewing, (2) boat and airplane traffic, (3) research activities, (4) timber harvest, (5) hard mineral extraction, (6) oil and gas exploration, (7) coastal development and pollutant discharge, and others.

During 2016 to 2017, SURTASS LFA sonar will not operate in the Gulf of Alaska. We expect any SURTASS LFA sonar signals that may propagate within proximity of the Steller sea lion designated critical habitat (closest mission area is Hawaii North) would have been reduced to negligible sound levels due to transmission loss. Therefore, the potential effect of SURTASS LFA sonar on Steller sea lion designated critical habitat is insignificant. We conclude that the SURTASS LFA sonar training, testing, and military operations the U.S. Navy proposes to conduct in the action area for the period of August 15, 2016 through August 14, 2017 pursuant to the five-year MMPA rule (August 15, 2012 through August 14, 2017) and ongoing for the reasonably foreseeable future will not destroy or adversely modify the critical habitat that has been designated for Steller sea lions. As a result, designated critical habitat of Steller sea lions will not be considered further in this opinion and conference report.

4.2 ESA-listed Species Likely to be Adversely Affected

This Section examines the status of each species that may be adversely affected by SURTASS LFA sonar activities. 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. 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), among others.

4.2.1 Blue Whale

The blue whale, *Balaenoptera musculus*, is a cosmopolitan species of baleen whale. It is the largest animal ever known to have lived on Earth: adults in the Antarctic have reached a maximum body length of about 33 m (108 ft) and can weigh more than 150,000 kg (330,700 lbs). The largest blue whales reported from the North Pacific are a female that measured 26.8 m (88 ft) taken at Port Hobron in 1932 (Reeves et al. 1985) and a 27.1 m (89 ft) female taken by Japanese pelagic whaling operations in 1959 (NMFS 1998b).

As is true of other baleen whale species, female blue whales are somewhat larger than males. Blue whales are identified by the following characteristics: a long-body and comparatively slender shape; a broad, flat "rostrum" when viewed from above; a proportionately smaller dorsal fin than other baleen whales; and a mottled gray color pattern that appears light blue when seen through the water. Blue whales may reach 70 to 80 years of age (COSEWIC 2002; Yochem and Leatherwood 1985a).

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 (IWC) 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 SURTASS LFA 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 and conference report, 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 1982). 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. 1982b; 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 1985a). 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 2003b; 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 1982).

Blue whales from both the eastern and western North Pacific have been heard, tracked, or harvested in waters off Kodiak Island; acoustic detections are made in the Gulf of Alaska from mid-July to mid-December and a peak from August through November (COSEWIC 2002; Ivashin and Rovnin. 1967; Moore et al. 2006; Stafford 2003a; Stafford et al. 2007; Yochem and Leatherwood 1985b). 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 2003a; Stafford et al. 2007; Stafford and Moore 2005a). However, surveys in the western Gulf of Alaska and east of Kodiak Island have not found blue whales (Rone et al. 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 2003a).

4.2.1.3 Reproduction

Gestation takes ten to 12 months, followed by a six to seven month nursing period. Sexual maturity occurs at five to 15 years of age and calves are born at two to three year intervals (COSEWIC 2002; NMFS 1998c; Yochem and Leatherwood 1985a). Recent data from illegal Russian whaling for Antarctic and pygmy blue whales support sexual maturity at 23 m (75 ft) and 19 to 20 m (62 to 65 ft), 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 per hour) while traveling versus while foraging (1.7 km per hour)(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 1998; Hucke-Gaete et al. 2004; Reilly and Thayer 1990; Yochem and Leatherwood 1985a). 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. 2004; Clapham et al. 1999a; 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 1985a). 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 2005b). 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 five to 20 times at 12 to 20 second intervals before a deep dive of three to 30 minutes (Croll et al. 1999a; Leatherwood et al. 1976; Maser et al. 1981; Yochem and Leatherwood 1985b). Average foraging dives are 140 m (459 ft) deep and last for 7.8 minutes (Croll et al. 2001a). Non-foraging dives are shallower and shorter, averaging 68 m (223 ft) and

4.9 minutes (Croll et al. 2001a). However, dives of up to 300 m (984 ft) are known (Calambokidis et al. 2003). Nighttime dives are generally shallower (50 m [164 ft]).

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).

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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 one to two 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: 1 μPa, but may reach 195 dB re: 1 μPa (Aburto et al. 1997; 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) at 1 m in the 17 to 30 Hz range and pygmy blue whale calls at 175± 1 dB re: 1 µPa (rms) at 1 m in the 17 to 50 Hz range.

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

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

Blue whale vocalizations tend to be long (greater than 20 s), low-frequency (less than 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 1971b; McDonald et al. 2001b) and 195 dB re: 1 μPa-m (Aburto et al. 1997) source levels. Calls are short-duration sounds (two to five seconds) 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 30 Hz), with seasonally variable occurrence.

Blue whale songs consist of repetitively patterned vocalizations produced over time spans of minutes to hours or even days (Cummings and Thompson 1971b; McDonald et al. 2001b). The songs are divided into pulsed/tonal units, which are continuous segments of sound, and phrases, repeated in combinations of one to five units (Mellinger and Clark 2003; Payne and McVay 1971). 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 approximately 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 seven of the world'sten 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 California, blues whales produce two predominant call types: Type B and D. B-calls are stereotypic of the blue whale population found in the eastern North Pacific (McDonald et al. 2006b) and are produced exclusively by males and associated with mating behavior (Oleson et al. 2007a). These calls have long durations (20 seconds) and low frequencies (10 to 100 Hz); they are produced either as repetitive sequences (song) or as singular calls. The B call has a set of harmonic tonals, and may be paired with a pulsed type A call. Blue whale D calls are down-swept in frequency (100 to 40 Hz) with duration of several seconds. These calls are similar worldwide and are associated with feeding animals; they may be produced as call-counter-call between multiple animals (Oleson et al. 2007b). In the SOCAL Range Complex region, D calls are produced in highest numbers during the late spring and early summer, and in diminished numbers during the fall, when A-B song dominates blue whale calling (Hildebrand et al. 2011; Hildebrand et al. 2012; Oleson et al. 2007c).

Calling rates of blue whales tend to vary based on feeding behavior. Stafford et al. (2005b) recorded the highest calling rates when blue whale prey was closest to the surface during its vertical migration. Wiggins et al. (2005) reported the same trend of reduced vocalization during daytime foraging followed by an increase at dusk as prey moved up into the water column and dispersed. Blue whales make seasonal migrations to areas of high productivity to feed, and vocalize less at the feeding grounds than during migration (Burtenshaw et al. 2004). Oleson et al. (2007c) reported higher calling rates in shallow diving (less than 30 m [100 ft]) whales, while deeper diving whales (greater than 50 m [165 ft]) were likely feeding and calling less.

Direct studies of blue whale hearing have not been conducted, but it is assumed that blue whales can hear the same frequencies that they produce (low-frequency) and are likely most sensitive to this frequency range (Ketten 1997b; Richardson et al. 1995d). Based on vocalizations and anatomy, blue whales are assumed to predominantly hear low-frequency sounds below 400 Hz (Croll et al. 2001b; Oleson et al. 2007c; 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. 2007a).

For discussion on the current understanding of how blue whales may respond to sonar, see Section 6.2.

4.2.1.8 Status and Trends

Blue whales (including all subspecies) were originally ESA-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.

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 (2010b) 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 three 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 PIFSC 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. 2011a), there have been no sightings of blue whales during systematic surveys off Hawaii (Barlow 2006; Mobley Jr. et al. 2000).

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. 1997; 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 U.S. Navy training and testing activities in the action 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 five 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 one 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*, 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 two to seven 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° South 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° North, 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° North throughout winter (Mizroch et al. 1999b). A resident fin whale population may exist in the Gulf of California (Tershy et al. 1993).

Fin whales are observed year-round off central and southern California with peak numbers in the summer and fall (Barlow 1997a; Campbell et al. 2015; Dohl et al. 1983; Forney et al. 1995). 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 1982). 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° North off the coast of California (Barlow 1995; Forney et al. 1995). 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 40.2 km per hour (25 miles 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 IWC 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 1966a; 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 1974b); and (5) a group centered in the Sea of Cortez (Gulf of California).

4.2.2.3 Reproduction

Fin whales reach sexual maturity between five and 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 six to 11 months (Boyd et al. 1999; Hain et al. 1992). The average calving interval in the North Atlantic is estimated at about two to three 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) 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, pregnancy 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 five to 20 shallow dives, each of 13 to 20 seconds duration, followed by a deep dive of 1.5 to 15 minutes (Gambell 1985a; Lafortuna et al. 2003; Stone et al. 1992). Other authors have reported that the fin whale’s most common dives last two to six minutes (Hain et al. 1992; Watkins 1981a). The most recent data support average dives of 98 m and 6.3 minutes for foraging fin whales, while non-foraging dives are 59 m (194 ft) and 4.2 minutes (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 (492 ft) are known (Panigada et al. 1999).

Fin whales along Southern California were found to be traveling 87 percent of the time and milling five 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 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 seconds) in the 18 to 35 Hz range, but only males are known to produce these (Clark et al. 2002; Patterson and Hamilton 1964). Richardson et al. (1995d) reported the most common sound as a one second vocalization of about 20 Hz, occurring in short series during spring, summer, and fall, and in repeated stereotyped patterns in winter. Au (Au and Green 2000b) reported moans of 14 to 118 Hz, with a dominant frequency of 20 Hz, tonal vocalizations of 34 to 150 Hz, and songs of 17 to 25 Hz (Cummings and Thompson 1994; Edds 1988; Watkins 1981b). Source levels for fin whale vocalizations are 140 to 200 dB re: 1 μPa-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 (164 ft) (Watkins et al. 1987).

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

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

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

Fin whales produce a variety of low frequency (less than 1 kHz) sounds, but the most typically recorded is a 20 Hz pulse lasting about one second, and reaching source levels of 189 ± 4 dB re: 1 μPam (Charif et al. 2002; Clark et al. 2002; Edds 1988; Richardson et al. 1995d; Sirovic et al. 2007; Watkins 1981b; Watkins et al. 1987). These pulses frequently occur in long sequenced patterns, are down swept (e.g., 23 to 18 Hz), and can be repeated over the course of many hours (Watkins et al. 1987). In temperate waters, intense bouts of these patterned sounds are very common from fall through spring, but also occur to a lesser extent during the summer in high latitude feeding areas (Clarke and Charif 1998). The seasonality and stereotypic nature of these vocal sequences suggest that they are male reproductive displays (Watkins 1981b; Watkins et al. 1987); a notion further supported by recent data linking these vocalizations to male fin whales only (Croll et al. 2002). In Southern California, the 20 Hz pulses are the dominant fin whale call type associated both with call-counter-call between multiple animals and with singing (Navy 2010; Navy 2012b). An additional fin whale sound, the 40 Hz call described by Watkins (1981b), was also frequently recorded, although these calls are not as common as the 20 Hz fin whale pulses. Seasonality of the 40 Hz calls differed from the 20 Hz calls, since 40 Hz calls were more prominent in the spring, as observed at other sites across the northeast Pacific (Sirovic et al. 2012). Source levels of Eastern Pacific fin whale 20-Hz calls has been reported as 189 +/- 5.8 dB re 1uPa at 1m (Weirathmueller et al. 2013). Although acoustic recordings of fin whales from many diverse regions show close adherence to the typical 20 Hz bandwidth and sequencing when performing these vocalizations, there have been slight differences in the pulse patterns, indicative of some geographic variation (Thompson et al. 1992; Watkins et al. 1987).

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

Although no studies have directly measured the sound sensitivity of fin whales, experts assume that fin whales are able to receive sound signals in roughly the same frequencies as the signals they produce. This suggests fin whales, like other baleen whales, are more likely to have their best hearing capacities at low frequencies, including frequencies lower than those of normal human hearing, rather than at mid- to high-frequencies (Ketten 1997b). Several fin whales were tagged during the Southern California Behavioral Response Study (SOCAL BRS) 2010 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. 2007a).

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 throughout the rest of their range (NMFS 2011b).

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

Fin whales are typically not expected south of 20° North 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° North, and significantly more sightings north of 40° North. However, they also showed limited search effort south of 20° North. The PIFSC 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).

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 per hour) 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 2005b). 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 2005b). In the Antarctic Ocean, fin whales are hunted by Japanese whalers who have been allowed to kill up to ten 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. 2007a). 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. (2007a), 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. 2005; Nelson et al. 2007). 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 2003b).

The vast majority of ship strike mortalities are never identified, and actual mortality is higher than currently documented. It is U.S. 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. 2012).

The organochlorines dichlorodiphenyldichloroethylene (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 Western North Pacific Gray Whale

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

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

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

4.2.3.1 Distribution

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

4.2.3.2 Population Structure

Gray whales occur in two genetically and spatially distinct populations on the eastern and western sides of the North Pacific Ocean (Brownell Jr. et al. 2009; Burdin et al. 2011; Kanda et al. 2010; Lang et al. 2004; Lang et al. 2005; Lang et al. 2010b; Leduc et al. 2002; Swartz et al. 2006; Weller et al. 2007; Weller et al. 2004b; Weller et al. 2006a).

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

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

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

4.2.3.3 Natural Threats

Predation by killer whales is a significant threat to gray whales, with calves being particularly susceptible during their northward migration (Fay et al. 1978; Goley and Straley 1994; Poole 1984; Rice and Wolman 1971; Ternullo and Black 2002). However, not all attacks are fatal and many individuals escape with scars from the encounters. Killer whales are frequently observed in the primary western North Pacific gray whale foraging area and roughly one-third to one half of observed gray whales bear tooth marks from killer whales (30 percent of them during the course of the ten year study) (Bradford et al. 2003; Bradford et al. 2006a; Weller et al. 2009a; Weller et al. 2002a). Vladimirov (2005) documented an attack on a mother-calf pair in shallow waters of the Piltun Bay foraging area. This rate is among the highest rate found amongst baleen whales. Researchers have also expressed significant concern about whales that appear “skinny”; the cause and consequences remain unknown (Bradford et al. 2007; Bradford et al. 2008; Bradford et al. 2012b; Burdin et al. 2003; IWC 2003; Weller et al. 2008b; Weller et al. 2007; Weller et al. 2005; Weller et al. 2004a; Weller et al. 2004b; Weller et al. 2006a; Yakovlev and Tyurneva 2004). Lactating females appear to be in particularly poor body condition (Bradford et al. 2012b).

4.2.3.4 Anthropogenic Threats

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

4.2.3.5 Status and Trends

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

Alter et al. (2007) concluded that eastern and western North Pacific gray whales historically numbered between 76,000 and 118,000 individuals combined prior to whaling; the proportion of individuals that was in each population is unknown. However, whaling dramatically reduced the population to a tiny fraction of its former abundance, with 100 to 130 non-calves remaining (Burdin et al. 2010; Cooke et al. 2005; Cooke et al. 2008; Reeves et al. 2008; Wade et al. 2003a; Weller et al. 2005; Weller et al. 2006a). The most recent abundance estimate of the western North Pacific gray whale population is 140 individuals (Carretta et al. 2015, unpublished 2014 stock assessment reports). The population was believed to be extinct in the 1970's (Bradford et al. 2003). At least 1,700 to 2,000 individuals were commercially harvested from the late 1800's to the mid-20th century (Commission 2004; IWC 2003). Findings that eastern North Pacific gray whales may be found within the range of western North Pacific gray whales may mean that even fewer individuals compose the western population, as individuals formerly believed to be western individuals may actually be part of the eastern population (Lang et al. 2010b).

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

4.2.3.6 Vocalizations and Hearing

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

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

4.2.3.7 Critical Habitat

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

4.2.4 Humpback Whale

Humpback whales (*Megaptera novaeangliae*) are distinguished from other whales in the same Family (Balaenopteridae) by extraordinarily long flippers (up to 5 m or about 1/3 total body length), a more robust body, fewer throat grooves (14 to 35), more variable dorsal fin, and utilization of very long (up to 30 minutes), complex, repetitive vocalizations (songs) (Payne and McVay 1971) during courtship. Their grayish-black baleen plates, approximately 270 to 440 on each side of the jaw, are intermediate in length (6,570 cm) to those of other baleen whales. Humpbacks in different geographical areas vary somewhat in body length, but maximum recorded size is 18 m (Winn and Reichley 1985a).

The whales are generally dark on the back, but the flippers, sides and ventral surface of the body and flukes may have substantial areas of natural white pigmentation plus acquired scars (white or black). Researchers distinguish individual humpbacks by the apparently unique black and white patterns on the underside of the flukes as well as other individually variable features (Glockner and Venus 1983; Katona and Whitehead 1981; Kaufman and Osmond 1987).

4.2.4.1 Distribution

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

In the North Pacific Ocean, the summer range of humpback whales includes coastal and inland waters from Point Conception, California, north to the Gulf of Alaska and the Bering Sea, and west along the Aleutian Islands to the Kamchatka Peninsula and into the Sea of Okhotsk (Tomlin 1967, Nemoto 1957, Johnson and Wolman 1984 as cited in NMFS 1991a). These whales migrate to calving grounds near Hawaii, southern Japan, the Mariana Islands, and Mexico during the winter months.

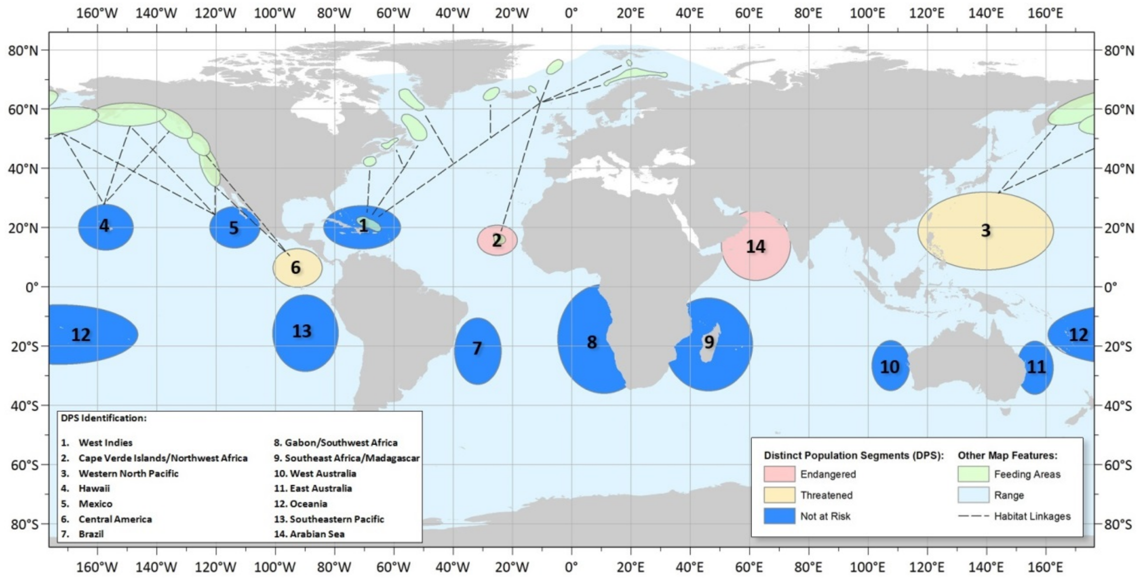

Figure 6. Worldwide distribution of the 14 identified humpback DPSs (80 FR 22304).

4.2.4.2 Population Structure

Though the ESA-listed entity is the worldwide population of humpback whales, evidence suggests there is likely multiple distinct populations within the North Pacific Ocean (e.g., see section 4.2.4.7). Descriptions of the population structure of humpback whales differ depending on whether an author focuses on where humpback whales winter or where they feed. During winter months in northern or southern hemispheres, adult humpback whales migrate to specific areas in warmer, tropical waters to reproduce and give birth to calves. During summer months, humpback whales migrate to specific areas in northern temperate or sub-arctic waters to forage. In summer months, humpback whales from different “reproductive areas” will congregate to feed; in the winter months, whales will migrate from different foraging areas to a discrete wintering area. In either case, humpback whales appear to form “open” populations; that is, populations that are connected through the movement of individual animals.

Based on genetic and photo-identification studies, four stocks have been recognized that likely correspond to populations of humpback whales in the North Pacific Ocean: two in the eastern North Pacific, one in the central North Pacific, and one in the western Pacific (Hill and DeMaster 1998a). However, gene flow between them may exist. Humpback whales summer in coastal and inland waters from Point Conception, California, north to the Gulf of Alaska and the Bering Sea, and west along the Aleutian Islands to the Kamchatka Peninsula and into the Sea of Okhotsk (Johnson and Wolman 1984; Nemoto 1957; Tomilin 1967). These whales migrate to Hawaii, southern Japan, the Mariana Islands, and Mexico during winter. However, more northerly penetrations in Arctic waters occur on occasion (Hashagen et al. 2009). Based on genetic and photo-identification studies, Hill and DeMaster (1998a) recognized four stocks, likely corresponding to populations of humpback whales in the North Pacific Ocean: two in the eastern North Pacific, one in the central North Pacific, and one in the western Pacific (Hill and DeMaster 1998a). However, gene flow between them may exist. The central North Pacific population winters in the waters around Hawaii while the eastern North Pacific population (also called the California-Oregon-Washington stock) winters along Central America and Mexico. However, Calambokidis et al. (1997) identified individuals from several populations wintering (and potentially breeding) in the areas of other populations, highlighting the potential fluidity of population structure. Herman (1979) presented extensive evidence that humpback whales associated with the MHIs immigrated there only in the past 200 years. Winn and Reichley (1985a) identified genetic exchange between the humpback whales that winter off Hawaii and Mexico (with further mixing on feeding areas in Alaska) and suggested that humpback whales that winter in Hawaii may have emigrated from Mexican wintering areas. A group of humpback whales winters in the South China Sea east through the Philippines, Ryukyu Retto, Ogasawara Gunto, Mariana Islands, and Marshall Islands, with occurrence in the Mariana Islands, at Guam, Rota, and Saipan from January through March (Darling and Cerchio 1993; Eldredge 1991; Eldredge 2003; Fulling et al. 2011; Rice 1998b). During summer, whales from this population migrate to the Kuril Islands, Bering Sea, Aleutian Islands, Kodiak, Southeast Alaska, and British Columbia to feed (Angliss and Outlaw 2008; Calambokidis 1997; Calambokidis et al. 2001).

Humpback whales primarily feed along the shelf break and continental slope (Green et al. 1992; Tynan et al. 2005b). Separate feeding groups of humpback whales are thought to inhabit western U.S. and Canadian waters, with the boundary between them located roughly at the U.S./Canadian border. The southern feeding ground ranges between 32 and 48° North, with limited interchange with areas north of Washington State (Calambokidis et al. 2004; Calambokidis et al. 1996).

4.2.4.3 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 one 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 example of male dominance polygamy (Clapham 1996a). Calving occurs in the shallow coastal waters of continental and insular shelves 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 to 100 years (COSEWIC 2011).

4.2.4.4 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. 1982a; Hain et al. 1995b; Jurasz and Jurasz 1979; Weinrich et al. 1992a; Witteveen et al. 2011). The principal fish prey in the western North Atlantic are sand lance, herring, and capelin (Kenney et al. 1985). There is good evidence of some territoriality on feeding and calving areas (Clapham 1994a; Clapham 1996a; Tyack 1981b). 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).

4.2.4.5 Diving and Social Behavior

In Hawaiian waters, humpback whales remain almost exclusively in waters within the 1,800 m (5,906 ft) isobath and usually within water depths less than 182 m (597 ft). Maximum diving depths are approximately 170 m (558 ft) (but usually less than 60 m [197 ft]), with a very deep dive (240 m [787 ft]) recorded off Bermuda (Hamilton et al. 1997a). Dives can last for up to 21 minutes, although feeding dives ranged from 2.1 to 5.1 minutes in the North Atlantic (Dolphin 1987a). In southeast Alaska, average dive times were 2.8 minutes for feeding whales, 3.0 minutes for non-feeding whales, and 4.3 minutes for resting whales (Dolphin 1987a). In the Gulf of California, humpback whale dive durations averaged 3.5 minutes (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, the primary prey of humpbacks, are found primarily between 92 and 120 m (302 to 394 ft); 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 1985c). Humpback whales wintering in the West Indies migrate relatively directly to the Gulf of Maine and areas around Iceland and Norway (Kennedy et al. 2013). 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 two to four individuals, although larger groups are seen near Shuyak and Sitkalidak islands and groups of 20 or more have been documented (Wynne et al. 2005).

Humpback whale feeding occurs in high latitudes during summer months. They exhibit a wide range of foraging behaviors and feed on a range of prey types, including small schooling fishes, krill, and other large zooplankton. In a review of humpback whale social behavior, Clapham (1996b) reported that they form small, unstable social groups during the breeding season and form small groups that occasionally aggregate on concentrations of food during the feeding season. The breeding season can best be described as a floating lek or male dominance polygyny (Clapham 1996b).

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. 1982b; Hain et al. 1995a; Jurasz and Jurasz 1979; Weinrich et al. 1992b). There is good evidence of some territoriality on feeding and calving areas (Clapham 1994b; Clapham 1996a; Tyack 1981a). Humpback whales are generally believed to fast while migrating and on breeding grounds, but some individuals apparently feed while in low-latitude waters normally believed to be used exclusively for reproduction and calf-rearing (Danilewicz et al. 2009; Pinto De Sa Alves et al. 2009). Some individuals, such as juveniles, may not undertake migrations at all (Findlay and Best. 1995).

Humpback whales feed on pelagic schooling euphausiids and small fish including capelin, herring and mackerel. Like other large mysticetes, they are a “lunge feeder” taking advantage of dense prey patches and engulfing as much food as possible in a single gulp. They also blow nets, or curtains, of bubbles around or below prey patches to concentrate the prey in one area, then lunge with open mouths through the middle. Dives appear to be closely correlated with the depths of prey patches, which vary from location to location. In the north Pacific (southeast Alaska), most dives were of fairly short duration (less than four minutes) with the deepest dive to 148 m (486 ft) (Dolphin 1987b), while whales observed feeding on Stellwagen Bank in the North Atlantic dove to less than 40 m (131 ft) (Hain et al. 1995a). Hamilton et al. (1997b) tracked one possibly feeding whale near Bermuda to 240 m (787 ft) depth.

4.2.4.6 Vocalization and Hearing

Humpback whale vocalization is much better understood than is hearing. Different sounds are produced that correspond to different functions: feeding, breeding, and other social calls (Dunlop et al. 2008). Males sing complex sounds while in low-latitude breeding areas in a frequency range of 20 Hz to 4 kHz with estimated source levels from 144 to 174 dB (Au et al. 2006a; Au et al. 2000b; Frazer and Mercado III 2000; Richardson et al. 1995d; Winn et al. 1970). Males also produce sounds associated with aggression, which are generally characterized by frequencies between 50 Hz to 10 kHz with most energy below 3 kHz (Silber 1986; Tyack 1983). Such sounds can be heard up to 9 km away (Tyack 1983). Other social sounds from 50 Hz to 10 kHz (most energy below 3 kHz) are also produced in breeding areas (Richardson et al. 1995d; Tyack 1983). While in northern feeding areas, both sexes vocalize in grunts (25 Hz to 1.9 kHz), pulses (25 to 89 Hz) and songs (ranging from 30 Hz to 8 kHz but dominant frequencies of 120 Hz to 4 kHz), which can be very loud (175 to 192 dB re: 1 μPa at 1 m) (Au et al. 2000b; Erbe 2002a; Payne 1985; Richardson et al. 1995d; Thompson et al. 1986). However, humpbacks tend to be less vocal in northern feeding areas than in southern breeding areas (Richardson et al. 1995d). NMFS classifies humpback whales in the low-frequency cetacean (i.e., baleen whale) functional hearing group. As a group, it is estimated that baleen whales can hear frequencies between 0.007 and 30 kHz (NOAA 2013). Houser et al. (2001b) produced a mathematical model of humpback whale hearing sensitivity based on the anatomy of the humpback whale ear. Based on the model, they concluded that humpback whales would be sensitive to sound in frequencies ranging from 0.7 to 10 kHz, with a maximum sensitivity between two to six kHz.

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 and sung only by adult males (Clark and Clapham 2004; Gabriele and Frankel. 2002; Helweg et al. 1992; Schevill et al. 1964; Smith et al. 2008). Singing is most common on breeding grounds during the winter and spring months, but is occasionally heard in other regions and seasons (Clark and Clapham 2004; Gabriele and Frankel. 2002; McSweeney et al. 1989). Au et al. (2000a) noted that humpbacks off Hawaii tended to sing louder at night compared to the day. There is geographical variation in humpback whale song, with different populations singing a basic form of a song that is unique to their own group. However, the song evolves over the course of a breeding season but remains nearly unchanged from the end of one season to the start of the next (Payne et al. 1983). The song is an elaborate series of patterned vocalizations that are hierarchical in nature, with a series of songs ('song sessions') sometimes lasting for hours (Payne and McVay 1971). Components of the song range from below 20 Hz up to 4 kHz, with source levels measured between 151 and 189 dB re: 1 μPa-m and high-frequency harmonics extending beyond 24 kHz (Au et al. 2006a; Winn et al. 1970).

Social calls range from 20 Hz to 10 kHz, with dominant frequencies below 3 kHz (D'Vincent et al. 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 one 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[7]) (Stimpert et al. 2007). Underwater lunge behavior was associated with nocturnal feeding at depth and with multiple bouts of broadband click trains that were acoustically different from toothed whale echolocation: Stimpert et al. (Stimpert et al. 2007) termed these sounds "mega-clicks" which showed relatively low received levels at the DTAGs (143 to 154 dB re: 1 μPa), with the majority of acoustic energy below 2 kHz.

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. 2007a). 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). Research by Au et al. (2001) and Au et al. (2006b) off Hawaii indicated the presence of high-frequency harmonics in vocalizations up to and beyond 24 kHz. While recognizing this was the upper limit of the recording equipment, it does not demonstrate that humpbacks can actually hear those harmonics, which may simply be correlated harmonics of the frequency fundamental in the humpback whale song. The ability of humpbacks to hear frequencies around 3 kHz may have been demonstrated in a playback study. Maybaum (1990)

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

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: 1 μPa-m or frequency sweep of 3.1 to 3.6 kHz. 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.

4.2.4.7 Status and Trends

Humpback whales were originally listed under the ESA as endangered in 1970 (35 FR 18319). On April 21, 2015, NMFS proposed to divide the globally listed endangered humpback whale population into 14 DPSs (80 FR 22304). Ten of these DPSs are not proposed for listing under the ESA, while two are proposed as endangered and two are proposed as threatened. The humpback whales in the western North Pacific part of the SURTASS LFA sonar action area belong to the proposed threatened Western North Pacific DPS; humpback whales in the central North Pacific part of the action area for SURTASS LFA sonar are part of the proposed Hawaii DPS of humpback whales, which is not proposed for listing under the ESA; humpback whales in the Indian Ocean part of the action area for SURTASS LFA sonar are part of the proposed endangered Arabian Sea DPS and West Australia DPS, which is not proposed for listing under the ESA.

The humpback whale is endangered because of past commercial whaling. Historical commercial harvests in the North Atlantic, North Pacific, and Southern Hemisphere are as follows:

- North Atlantic: approximately 31,000 whales killed since the 1600s (Smith and Reeves 2010)
- North Pacific: nearly 28,000 whales killed between 1905 and 1965 (Perry et al. 1999)
- Southern Hemisphere: more than 200,000 whales killed in the 20th century (Findlay 2001)

Whaling for subsistence purposes does still occur for humpback whales, though at a reduced level compared to historical levels. Since 1985, there have been 98 humpback whale “catches” for subsistence purposes; of those catches, 59 were hunted by Denmark in Greenland and 38 were hunted by St. Vincent and the Grenadines in the West Indies (IWC 2015).

Winn and Reichley (1985b) argued that the global humpback whale population consisted of at least 150,000 whales in the early 1900s, mostly in the Southern Ocean. In 1987, the global population of humpback whales was estimated at about 10,000 (NMFS 1987). Although this estimate is outdated, it appears that humpback whale numbers are increasing. Though whaling does continue, the number of whales killed has been significantly reduced.

Humpback whale pre-exploitation abundance cannot be known, but various estimates have been proposed:

- Global (though mostly representative of the Southern Ocean): at least 150,000 whales in the early 1900s (Winn and Reichley 1985d)

- North Atlantic: estimates range from 40,000 to 250,000 (Smith and Pike 2009)
- North Pacific: 15,000 humpback whales prior to 1905 (Rice 1978a)

Currently, there are over 60,000 humpback whales worldwide, occurring primarily in the North Atlantic, North Pacific, and Southern Hemisphere. Though abundance estimates are not available for all populations or stocks worldwide, estimates are available for some major groups:

- North Atlantic: total population between 7,698 and 11,570 (Palsbøll et al. 1997; Smith et al. 1999; Stevick et al. 2003), though all are considered to be under estimates
- North Pacific: total population is estimated to be 20,800 (Barlow et al. 2011a), though this is likely an underestimate
- Southern Hemisphere: total population estimated by IWC in 1997/98 was 42,000

Though all populations of humpback whales are depressed relative to pre-exploitation levels, population growth appears to be positive. Growth rates for populations worldwide vary between 3.1 to 10.0 percent (Allen and Angliss 2014b; Angliss and Outlaw 2005; Barlow 1997b; Barlow et al. 2011a; Calambokidis et al. 2008b; Hendrix et al. 2012; Katona and Beard 1990; Punt 2010a; Saracco et al. 2013; Stevick et al. 2003).

It is estimated that 15,000 humpback whales resided in the North Pacific in 1905 (Rice 1978b). However, from 1905 to 1965, nearly 28,000 humpback whales were harvested in whaling operations, reducing the number of all North Pacific humpback whale to roughly 1,000 (Perry et al. 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. 2011b). 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 ten years (Calambokidis et al. 2008a). The current (year 2008) best estimate of abundance for the California, Oregon, and Washington stock is 1,918 (coefficient of variation = 0.03), with an estimated mimimum population size estimate of 1,876 individuals (Carretta et al. 2014). Based on ship surveys conducted in the summer and fall from 1991 to 2005, it is estimated that 36 humpback whales (coefficient of variation = 0.51) occur off Southern California in the waters south of Point Conception (Barlow and Forney 2007). Humpback whale density estimates vary annually and by season. The average density of humpback whales off Southern California between 2004 and 2013 was 1.07, 1.92, 1.03, and 0.82 individuals per 1,000 km^2 in the winter, spring, summer, and fall, respectively (Campbell et al. 2014). The overall annual average density of humpback whales off Southern California over this time period was 1.17 individuals per km^2 (Campbell et al. 2014). The current (year 2006) best estimate of abundance for the Central North Pacific stock has been estimated at 10,103 individuals on wintering grounds throughout the main Hawaiian Islands (Allen and Angliss 2013a), with an estimated minimum population size of 5,833 individuals (Carretta et al. 2014).

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 Asia, an annual rate of growth of 6.7 percent has been estimated (Calambokidis et al. 2008a).

In the Pacific, the stock structure of humpback whales is defined based on feeding areas because of the species' fidelity to feeding grounds (Carretta et al. 2010). The NMFS has designated four stocks for management purposes under the MMPA: (1) the Central North Pacific stock, with feeding areas from Southeast Alaska to the Alaska Peninsula; (2) the Western North Pacific stock, with feeding areas from the Aleutian Islands, Bering Sea, and Russia; (3) the California, Oregon, Washington, and Mexico stock, with feeding areas off the U.S. west coast; and (4) the American Samoa stock, with feeding areas as far south as the Antarctic Peninsula (Allen and Angliss 2010b). Humpback whales in the western North Pacific part of the action area are most likely part of the Western North Pacific stock, since this population is known to winter in the Ogasawara Islands, Ryukyu (Okinawa) Islands, Taiwan, the Philippines, and the Mariana Islands (Calambokidis et al. 2008a). The current population estimate for this stock is 1,107 animals (Calambokidis et al. 2008a).

4.2.4.8 Natural Threats

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

Parasites and biotoxins from red-tide blooms are other potential causes of mortality (Perry et al. 1999). The occurrence of the nematode *Crassicauda boopis* appears to increase the potential for kidney failure in humpback whales and may be preventing some populations from recovering (Lambertsen 1992). Studies of 14 humpback whales that stranded along Cape Cod between November 1987 and January 1988 indicate they apparently died from a toxin produced by dinoflagellates during this period. One-quarter of humpback whales of the Arabian Sea population show signs of tattoo skin disease[8], which may reduce the fitness of afflicted individuals (Baldwin et al. 2010). Emaciated calf and juvenile humpbacks were found in numbers an order of magnitude higher than normal along Western Australia in 2009 due to unknown causes (Coughran et al. 2013).

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[8] irregular skin lesions of which the cause is unknown; disease is common in odontocetes but rare in baleen whales

4.2.4.9 Anthropogenic Threats

Human activities known or thought to threaten humpback whales include commercial fishing, shipping, and contaminants. Historically, whaling represented the greatest threat to every population of whales and was ultimately responsible for listing several species as endangered. However, with the exception of some subsistence harvest, this threat has largely been eliminated for humpback whales.

Humpback whales are also killed or injured during interactions with commercial fishing gear. Like fin whales, humpback whales have been entangled by fishing gear off Newfoundland and Labrador, Canada. A total of 595 humpback whales were reported captured in coastal fisheries in those two provinces between 1969 and 1990, of which 94 died (Lien 1994; Perkins and Beamish 1979). From 1979 through 2008, 1,209 whales were recorded entangled, 80 percent of which were humpback whales (Benjamins et al. 2012). Along the Atlantic coast of the U.S. and the Maritime Provinces of Canada, there were 160 reports of humpback whales being entangled in fishing gear between 1999 and 2005 (Cole et al. 2005; Nelson et al. 2007). Of these, 95 entangled humpback whales were confirmed, with 11 whales sustaining injuries and nine dying of their wounds. The first estimate of population-level effects of entanglement were recently produced, with over 12 percent of the Gulf of Maine population of humpbacks acquiring new scars from entanglement interactions annually (Mattila and Rowles 2010). Along the Pacific coast of Canada, 40 humpback whales have been reported as entangled since 1980, four of which are known to have died (COSEWIC 2011; Ford et al. 2009). Between 30 and 40 percent of humpback whales in the Arabian Sea show scarring from entanglements, with fishing effort on the rise (Baldwin et al. 2010). Alava et al. (2012) reported that 0.53 percent of humpback whale populations breeding along Ecuador are bycaught annually in commercial fishing gear (mortality of 15 to 33 individuals per year). From 2004 to 2008, 18 humpback whales were observed to be entangled along the U.S. west coast, of which 14 were considered seriously injured and two are known to have died (Carretta et al. 2013). From 2006 to 2010, 29 entangled whales were identified with serious injury or mortality resulting from the entanglement (Waring et al. 2013). From 1996 to 2000, 22 humpback whales of the Central North Pacific population were found entangled in fishing gear (Angliss and Lodge. 2004). In 1996, a vessel from the Pacific Missile Range Facility in Hawaii rescued an entangled humpback, removing two crabpot floats from the whale. A photography study of humpback whales in southeastern Alaska in 2003 and 2004 found at least 53 percent of individuals showed some kind of scarring from fishing gear entanglement (Neilson et al. 2005). There are also reports of entangled humpback whales from the Hawaiian Islands. In 1991, a humpback whale was observed entangled in longline gear and released alive (Hill et al. 1997). In 1995, a humpback whale in Maui waters was found trailing numerous lines (not fishery-related) and entangled in mooring lines. The whale was successfully released but subsequently stranded and was attacked and killed by tiger sharks in the surf zone. From 2001 through 2006, there were 23 reports of entangled humpback whales in Hawaiian waters; 16 of these reports were from 2005 and 2006. A photography study of humpback whales in southeastern Alaska in 2003 and 2004 found at least 53 percent of individuals showed some kind of scarring from fishing gear entanglement (Neilson et al. 2005). Between 30 and 40 percent of humpback whales in the Arabian Sea show scarring from entanglements, with fishing effort on the rise (Baldwin et al. 2010). Alava et al. (2012) reported that 0.53 percent of humpback whale populations breeding along Ecuador are bycaught annually in commercial fishing gear (mortality of 15 to 33 individuals per year). Ten humpback whales were found entangled in gill nets or long lines between 1995 and 2002 off Peru (Garcia-Godos et al. 2013).

Many of the entangled humpback whales observed in Hawaiian waters brought the gear with them from higher latitude feeding grounds; for example, in 1996, the whale the U.S. Navy rescued had been entangled in gear that was traced to a recreational fisherman in southeast Alaska. Thus far, 6 of the entangled humpback whales observed in the Hawaiian Islands have been confirmed to have been entangled in gear from Alaska. Nevertheless, humpback whales are also entangled in fishing gear in the Hawaiian Islands. Since 2001, there have been 5 observed interactions between humpback whales and gear associated with the Hawaii-based longline fisheries (NMFS 2008b). In each instance, however, all of the whales were disentangled and released or they were able to break free from the gear without reports of impairment of the animal's ability to swim or feed.

More humpback whales are killed in collisions with ships than any other whale species except fin whales (Jensen and Silber 2003a). Of 123 humpback whales that stranded along the Atlantic coast of the U.S. between 1975 and 1996, ten (8.1 percent) showed evidence of collisions with ships (Laist et al. 2001). From 1975 through 2011, 68 collisions were actually witnessed in the main Hawaiian Islands, 63 percent involving calves and subadults, with the rate of collisions increasing over time even accounting for higher numbers of whales present (Lammers et al. 2013). Between 1999 and 2005, there were 18 reports of humpback whales being struck by vessels along the Atlantic coast of the U.S. and the Maritime Provinces of Canada (Cole et al. 2005; Nelson et al. 2007). Of these reports, 13 were confirmed as ship strikes and in seven cases, ship strike was determined to be the cause of death. Along Pacific Canada, 21 reports of ship strikes involving humpback whales were reported from 2001 to 2008 (COSEWIC 2011; Ford et al. 2009). From 2006 to 2010, 10 instances of mortality stemming from vessel collision were documented (Waring et al. 2013). 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 ten knots and changing shipping lanes by less than one nmi to avoid the greatest concentrations of right whales are expected to reduce the chance of humpback whales being hit by ships by nine percent. Two humpbacks were recorded as ship strikes and died along the west coast from 2004 to 2008; a third was known to have been struck but its outcome is unknown (Carretta et al. 2013). On the Pacific coast, a humpback whale is killed about every other year by ship strikes (Barlow et al. 1997b). The humpback whale calf that was found stranded on Oahu with evidence of vessel collision (propeller cuts) in 1996 suggests that ship collisions might kill adults, juvenile, and calves (NMFS unpublished data). Although data for actual strikes is lacking off Pacific Panama, study of shipping data and satellite tag data on humpback whales showed that eight of 15 whales tagged came within 200 m of 81 different ships on 98 occasions in a period of 11 days (Guzman et al. 2013).

Organochlorines, including PCB and DDT, have been identified from humpback whale blubber (Gauthier et al. 1997a). Higher PCB levels have been observed in western Atlantic waters versus Pacific waters in United States waters and levels tend to increase with age (Elfes et al. 2010); eastern Atlantic individuals fall between these two in contaminant burden (Ryan et al. 2014). Although humpback whales in the Gulf of Maine and off Southern California tend to have the highest PCB concentrations, overall levels are on par with other baleen whales, which are generally lower than odontocete cetaceans (Elfes et al. 2010). As with other baleen whales, these contaminants are transferred to young through the placenta, leaving newborns with contaminant loads equal to that of mothers before bioaccumulating additional contaminants during life and passing the additional burden to the next generation (Metcalfe et al. 2004). Contaminant levels are relatively high in humpback whales compared to blue whales since humpback whales feed not only on zooplankton but also on fishes, which carry higher contaminant loads than the krill on which blue whales solely feed.

Foraging locations in the North Atlantic have shifted by significant distances over the past few decades, potentially as a result of global warming (Palsboll et al. 2013).

4.2.4.10 Critical Habitat

The NMFS has not designated critical habitat for humpback whales.

4.2.5 North Pacific Right Whale

4.2.5.1 Population Structure

All North Pacific right whales constitute a single population, although debate exists about subdivisions (Kennedy et al. 2012; Leduc et al. 2012).

4.2.5.2 Distribution and Abundance

Very little is known of the distribution of right whales in the North Pacific and very few of these animals have been seen in the past 20 years. Historical whaling records indicate that right whales ranged across the North Pacific north of 30° North latitude and occasionally as far south as 20° North, with a bimodal distribution longitudinally favoring the eastern and western North Pacific and occurring infrequently in the central North Pacific (Gregr and Coyle. 2009; Josephson et al. 2008a; Maury 1853; Scarff 1986; Scarff 1991; Townsend 1935a). North Pacific right whales summered in the North Pacific and southern Bering Sea from April or May to September, with a peak in sightings in coastal waters of Alaska in June and July (Klumov 1962; Maury 1852; Omura 1958; Omura et al. 1969a; Townsend 1935a). North Pacific right whale summer range extended north of the Bering Strait (Omura et al. 1969a). However, they were particularly abundant in the Gulf of Alaska from 145 to 151° West, and apparently concentrated in the Gulf of Alaska, especially south of Kodiak Islands and in eastern Aleutian Islands and southern Bering Sea waters (Berzin and Rovnin 1966b; Braham and Rice 1984).

Current information on the seasonal distribution of right whales is lacking. In the eastern North Pacific, this includes sightings over the middle shelf of the Bering Sea, Bristol Bay, Aleutian and Pribilof Islands (Goddard and Rugh 1998; Hill and DeMaster 1998b; Perryman et al. 1999; Wade et al. 2006b; Waite et al. 2003a). Some more southerly records also record occurrence along Hawaii, California, Washington, and British Columbia (Herman et al. 1980; Scarff 1986). However, records from Mexico and California may suggest historical wintering grounds in offshore southern North Pacific latitudes (Brownell et al. 2001a; Gregr and Coyle. 2009). Sightings of several western population groups were made several hundred kilometers southeast of the Kamchatka Peninsula, where foraging may occur (Sekiguchi et al. 2014).

According to the 2013 Recovery Plan (78 FR 34347) the North Pacific right whale likely numbers less than 500 individuals, with 400 belonging to the Sea of Okhotsk and approximately 100 throughout the rest of the North Pacific.

4.2.5.3 Growth and Reproduction

While no reproductive data are known for the North Pacific, studies of North Atlantic right whales suggest calving intervals of two to seven years and growth rates that are likely dependent on feeding success (Best et al. 2001; Burnell 2001; Cooke et al. 2001; Kenney 2002; Knowlton et al. 1994; Reynolds et al. 2002). It is presumed that right whales calve during mid-winter (Clapham et al. 2004a). Western North Pacific sightings have been recorded along Japan, in the Yellow Sea, and Sea of Japan (Best et al. 2001; Brownell et al. 2001a). Lifespans of up to 70 years can be expected based upon North Atlantic right whale data.

4.2.5.4 Feeding

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

4.2.5.5 Habitat

Habitat preference data are sparse for North Pacific right whales as well. Sightings have been made with greater regularity in the western North Pacific, notably in the Okhotsk Sea, Kuril Islands, and adjacent areas (Brownell et al. 2001a). In the western North Pacific, feeding areas occur in the Okhotsk Sea and adjacent waters along the coasts of Kamchatka and the Kuril Islands (IWC 2001).

Historical concentrations of sightings in the Bering Sea together with some recent sightings indicate that this region, together with the Gulf of Alaska, may represent an important summer habitat for eastern North Pacific right whales (Brownell et al. 2001a; Clapham et al. 2004a; Goddard and Rugh 1998; Scarff 1986; Shelden et al. 2005a). North Pacific right whale occurrence in the Bering Sea during summer appears to be strongly influenced by the occurrence and abundance of the copepod *Calanus marshallae* (Baumgartner et al. 2013). Few sighting data are available from the eastern North Pacific, with a single sighting of 17 individuals in the southeast Bering Sea being by far the greatest known occurrence (Wade et al. 2006a). Some further sightings have occurred in the northern Gulf of Alaska (Wade et al. 2006a). Recent eastern sightings tend to occur over the continental shelf, although acoustic monitoring has identified whales over abyssal waters (Mellinger et al. 2004b). It has been suggested that North Pacific right whales have shifted their preferred habitat as a result of reduced population numbers, with oceanic habitat taking on a far smaller component compared to shelf and slope waters (Shelden et al. 2005b). The area where North Pacific right whales are densest in the Gulf of Alaska is between 150 and 170° West and south to 52° North (Shelden and Clapham 2006), but present occurrence there is very rare (Wade et al. 2011). However, four sightings were made from 2004 to 2006 off Kodiak Island in association with high zooplankton concentrations (Wade et al. 2011). A right whale was sighted southeast of Kodiak Island in July 1998 and acoustic detections have been made off Kodiak Island, although no detections occurred from April to August 2003 or in April 2009 (Munger et al. 2008; Rone et al. 2010; Waite et al. 2003b). The greatest frequency of call occurrence in the southeastern Bering Sea occurs from July to October (Munger et al. 2008).

4.2.5.6 Migration and Movement

Historical sighting and catch records provide the only information on possible migration patterns for North Pacific right whales (Omura 1958; Omura et al. 1969a; Scarff 1986). During summer, whales have been found in the Gulf of Alaska, along both coasts of the Kamchatka Peninsula, the Kuril Islands, the Aleutian Islands, the southeastern Bering Sea, and in the Okhotsk Sea. Fall and spring distribution was the most widely dispersed, with whales occurring in mid-ocean waters and extending from the Sea of Japan to the eastern Bering Sea. In winter, right whales have been found in the Ryukyu Islands (south of Kyushu, Japan), the Bonin Islands, the Yellow Sea, and the Sea of Japan. Whalers never reported winter calving areas in the North Pacific and where calving occurs remains unknown (Clapham et al. 2004a; Gregr and Coyle. 2009; Scarff 1986). North Pacific right whales probably migrate north from lower latitudes in spring and may occur throughout the North Pacific from May through August north of 40º North from marginal seas to the Gulf of Alaska and Bering Sea, although absence from the central North Pacific has been argued due to inconsistencies in whaling records (Clapham et al. 2004b; Josephson et al. 2008b). This follows generalized patterns of migration from high-latitude feeding grounds in summer to more temperate, possibly offshore waters, during winter (Braham and Rice 1984; Clapham et al. 2004a; Scarff 1986).

4.2.5.7 Vocalization and Hearing

Right whales vocalize to communicate over long distances and for social interaction, including communication apparently informing others of prey path presence (Biedron et al. 2005; Tyson and Nowacek 2005). Vocalization patterns amongst all right whale species are generally similar, with six major call types: scream, gunshot, blow, up call, warble, and down call (McDonald and Moore 2002; Parks and Tyack 2005). A large majority of vocalizations occur in the 300 to 600 Hz range with up- and down sweeping modulations (Vanderlaan et al. 2003). Vocalizations below 200 Hz and above 900 Hz were rare (Vanderlaan et al. 2003). Calls tend to be clustered, with periods of silence between clusters (Vanderlaan et al. 2003). Gunshot bouts last 1.5 hours on average and up to seven hours (Parks et al. 2012a). Blows are associated with ventilation and are generally inaudible underwater (Parks and Clark 2007). Up calls are 100 to 400 Hz (Gillespie and Leaper 2001). Gunshots appear to be a largely or exclusively male vocalization (Parks et al. 2005b).

Smaller groups vocalize more than larger groups and vocalization is more frequent at night (Matthews et al. 2001). Moans are usually produced within 10 m (33 ft) of the surface (Matthews et al. 2001). Up calls were detected year-round in Massachusetts Bay except July and August and peaking in April (Mussoline et al. 2012). Individuals remaining in the Gulf of Maine through winter continue to call, showing a strong diel pattern of up call and gunshot vocalizations from November through January possibly associated with mating (Bort et al. 2011; Morano et al. 2012; Mussoline et al. 2012). Estimated source levels of gunshots in non-surface active groups are 201 dB re: 1 μPa peak-to-peak (Hotchkin et al. 2011). While in surface active groups, females produce scream calls and males produce up calls and gunshot calls as threats to other males; calves (at least female calves) produce warble sounds similar top their mothers' screams (Parks et al. 2003; Parks and Tyack 2005). Source levels for these calls in surface active groups range from 137 to 162 dB re: 1 μPa-m (rms), except for gunshots, which are 174 to 192 dB re: 1 μPa-m (rms) (Parks and Tyack 2005). Up calls may also be used to reunite mothers with calves (Parks and Clark 2007). Atlantic right whales shift calling frequencies, particularly of up calls, as well as increase call amplitude over both long and short term periods due to exposure to vessel noise (Parks and Clark 2007; Parks et al. 2005a; Parks et al. 2007b; Parks et al. 2011; Parks et al. 2010; Parks et al. 2012b; Parks et al. 2006), particularly the peak frequency (Parks et al. 2009).

North Atlantic right whales respond to anthropogenic sound designed to alert whales to vessel presence by surfacing (Nowacek et al. 2003; Nowacek et al. 2004c).

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

4.2.5.8 Status and Trends

The Northern right whale was originally ESA-listed as endangered in 1970 (35 FR 18319), and this status remained since the inception of the ESA in 1973. The early listing included both the North Atlantic and the North Pacific populations, although subsequent genetic studies conducted by Rosenbaum (2000) resulted in strong evidence that the North Atlantic and North Pacific right whales are separate species. Following a comprehensive status review, NMFS concluded that Northern right whales are indeed two separate species. In March 2008, NMFS published a final rule listing North Pacific and North Atlantic right whales as separate species (73 FR 12024).

Very little is known about right whales in the eastern North Pacific, which were severely depleted by commercial whaling in the 1800s (Brownell et al. 2001a). At least 11,500 individuals were taken by American whalers in the early- to mid-19th century, but harvesting continued into the 20th century (Best 1987). Illegal Soviet whaling took 661 individuals between 1962 and 1968, with 529 from the eastern North Pacific and 152 from the Okhotsk Sea, mostly of large mature individuals (Brownell et al. 2001a; Ivashchenko and Clapham 2012; Ivashchenko et al. 2013). In the last several decades there have been markedly fewer sightings due to a drastic reduction in number, caused by illegal Soviet whaling in the 1960s (Doroshenko 2000). The current population size of right whales in the North Pacific is likely fewer than 1,000 animals compared to possibly 11,000 individuals or more prior to exploitation (NMFS 1991b; NMFS 2006e). Based upon mark-recapture estimates of abundance suggested right whales in the Bering Sea and Aleutian Islands numbered 31 individuals (95 percent CL 23 to 54, CV = 0.22) and 28 (95 percent CL 24 to 42), respectively and composed of eight females and 20 males (Wade et al. 2011).

Abundance estimates and other vital rate indices in both the eastern and western North Pacific are not well established. Where such estimates exist, they have very wide confidence limits. Previous estimates of the size of the right whale population in the Pacific Ocean range from a low of 100 to 200 to a high of 220 to 500 (Berzin and Yablokov 1978; Braham and Rice 1984). Although Hill and DeMaster (1998b) argued that it is not possible to reliably estimate the population size or trends of right whales in the North Pacific, Reeves et al. (2003) and Brownell et al. (2001) concluded that North Pacific right whales in the eastern Pacific Ocean exist as a small population of individuals while the western population of right whales probably consists of several hundred animals, although Clapham et al. (2005) placed this population at likely under 100 individuals (Wade et al. (2010) estimated 25 to 38 individuals). Brownell et al. (2001b) reviewed sighting records and also estimated that the abundance of right whales in the western

North Pacific was likely in the low hundreds. From 2007 to 2010, 12 individuals were observed in the southeastern Bering Sea (some on multiple occasions)(Allen and Angliss 2013b). Genetic analyses indicate genetic diversity to be low, but not as low as North Atlantic right whales (Leduc et al. 2012), and higher than what might be expected from such a small population (Slikas et al. 2013). However, genetic diversity in the next generation is expected to be severely reduced (Slikas et al. 2013).

Scientists participating in a recent study utilizing acoustic detection and satellite tracking identified 17 right whales (ten males and seven females) in the Bering Sea, which is almost threefold the number seen in any previous year in the last four decades (Wade et al. 2006b). These sightings increased the number of individual North Pacific right whales identified in the genetic catalog for the eastern Bering Sea to 23. Amidst the uncertainty of the eastern North Pacific right whale's future, the discovery of females and calves gives hope that this endangered population may still possess the capacity to recover (Wade et al. 2006b). Available age composition of the North Pacific right whale population indicates most individuals are adults (Kenney 2002). Length measurements for two whales observed off California suggest at least one of these whales was not yet sexually mature and two calves have been observed in the Bering Sea (Carretta et al. 1994; Wade et al. 2006b). However, to date, there is no evidence of reproductive success (i.e., young reared to independence) in the eastern North Pacific. No data are available for the western North Pacific. According to the 2013 Recovery Plan (78 FR 34347) the North Pacific right whale likely numbers less than 500 individuals, with 400 belonging to the Sea of Okhotsk and approximately 100 throughout the rest of the North Pacific.

4.2.5.9 Natural Threats

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

4.2.5.10 Anthropogenic Threats

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

Climate change may have a dramatic effect on survival of North Pacific right whales. Right whale life history characteristics make them very slow to adapt to rapid changes in their habitat (see Reynolds et al. 2002). They are also feeding specialists that require exceptionally high densities of their prey (see Baumgartner et al. 2003; Baumgartner and Mate 2003). Zooplankton abundance and density in the Bering Sea has been shown to be highly variable, affected by climate, weather, and ocean processes and in particular ice extent (Baier and Napp 2003; Napp and G.L. Hunt 2001). The largest concentrations of copepods occurred in years with the greatest southern extent of sea ice (Baier and Napp 2003). It is possible that changes in ice extent, density, and persistence may alter the dynamics of the Bering Sea shelf zooplankton community and in turn affect the foraging behavior and success of right whales. No data are available for the western North Pacific.

4.2.5.11 Critical Habitat

Designated critical habitat for North Pacific right whales is discussed in Section 4.1.5.

4.2.6 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 12 to 18 m (40 to 60 ft) and weigh 45,000 kg (100,000 lbs). 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 whale is regarded as the fastest swimmer among the great whales, reaching bursts of speed in excess of 37 km per hour (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 3 to 4 m (10 to 13 ft) 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 six to 12 years of age when they reach about 13 m (45 ft) in length, and generally mate and give birth during the winter in lower latitudes. Females breed every two to three years, with a gestation period of 11 to 13 months. Females give birth to a single calf that is about 4.6 m (15 ft) long and weighs about 680 kg (1,500 lbs). Calves are usually nursed for six to nine months before being weaned on the preferred feeding grounds. Sei whales have an estimated lifespan of 50 to 70 years.

4.2.6.1 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. 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 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° North (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 16.8º C. Sei whales have been seen in monitoring efforts in Hawaii in 2007 and in 2010.

4.2.6.2 Population Structure

The population structure of sei whales is not well defined but is 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° West, and another east of 155° West (Masaki 1976b; 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. 1982a; Nasu 1974b). Sightings have also occurred in Hawaiian waters. In U.S. 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 (10,171 to 14,764 ft). Sightings were made during Beaufort sea state two to four. 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° North (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 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 161 km (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).

4.2.6.3 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 six to nine months, and the calving interval is about two to three years (Gambell 1985c; Rice 1977). Sei whales become sexually mature at about age ten (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 to July 2009 cruises in the western North Pacific (Tamura et al. 2009).

4.2.6.4 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. 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 1985c).

4.2.6.5 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 1986a). The dominant food for sei whales off California during June to August is northern anchovy, while in September to 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, pollock, 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).

4.2.6.6 Diving and Social Behavior

Generally, sei whales make five to 20 shallow dives of 20 to 30 seconds 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 m. Sei whales are usually found in small groups of up to six individuals, but they commonly form larger groupings when they are on feeding grounds (Gambell 1985d).

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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.

4.2.6.7 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 seconds duration and tonal and upsweep calls in the 200 Hz to 600 Hz range of one to three seconds 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 seconds, separated by 0.4 to 1.0 seconds) of ten to 20 short (4 msec) FM sweeps between 1.5 to 3.5 kHz (Richardson et al. 1995d).

Recordings made in the presence of sei whales have shown that they produce sounds ranging from short, mid-frequency pulse sequences (Knowlton et al. 1991; Thompson et al. 1979a) 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 seconds and ranging in frequency from 1.5 to 3.5 kHz in the presence of sei whales—data similar to that reported by Thompson et al. (1979a). 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 seconds 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 82.3to 34.0 Hz and averaging 1.38 seconds 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 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.

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

4.2.6.8 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. 2008a) 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. 2011a). 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 to 23° North during the winter and from 35 to 50° North during the summer (Horwood 2009; Masaki 1976a; 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° North in the winter (Fulling et al. 2011). Observations during the U.S. Navy-funded 2007 survey of the action area indicated this species most often occurs in deep water 3,164 to9,322 m (10,381 to 30,583 ft). 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° North) 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. 2013). A sei whale was also detected with sonobuoys on the January to February 2010 *Oscar Elton Sette* Cruise from Hawaii to Guam. However, the reported information did not allow us to determine if this detection occurred in the SURTASS action area.

4.2.6.9 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).

4.2.6.10 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 U.S. 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. 2005; Nelson et al. 2007). 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 18.5 km per hour (10 knots) and changing shipping lanes by less than one nmi 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.

4.2.6.11 Critical Habitat

The NMFS has not designated critical habitat for sei whales.

4.2.7 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 11 m (36 ft) and weigh 13,607 kg (15 tons). Adult males, however, reach about 16 m (52 ft) and may weigh as much as 40,823 kg (45 tons).

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 7.8 kg [17 lb] 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.

4.2.7.1 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 North in the North Atlantic and 70º South 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 North or South (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.

4.2.7.2 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 with no clear geographic structure and 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 International Whaling Commission (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. 2004a). Genetic studies indicate that movements of both sexes through expanses of ocean basins are common, and that males, but not females, often breed in different ocean basins than the ones in which they were born (Whitehead 2003). Sperm whale populations appear to be structured socially at the level of the clan rather than geographically (Whitehead 2003; Whitehead 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} North in winter (Carretta et al. 2005; Stafford 2003b; Stafford et al. 2001a; Watkins et al. 2000). Sperm whales are found year-round in Californian and Hawaiian waters (Barlow 1995; Dohl et al. 1983; Forney et al. 1995; Lee 1993; Mobley Jr . et al. 2000; Rice 1960a; Shallenberger 1981), but they reach peak abundance from April through mid-June and from the end of August through mid-November (Rice 1974a). They are seen in every season except winter (December to 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. 2004a). Occurrence is higher from July through September than January through March (Mellinger et al. 2004a; Moore et al. 2006). The vast majority of individuals in the region are likely male based upon whaling records and genetic studies; the area is a summer foraging area for these individuals (Allen and Angliss 2010a; Reeves et al. 1985; Straley and O'Connell 2005; Straley et al. 2005). Mean group size has been reported to be 1.2 individuals (Wade et al. 2003b; Waite 2003). However, female groups may rarely occur at least up to the central Aleutian Islands (Fearnbach et al. 2012).

4.2.7.3 Movement

Mature males range between 70º North in the North Atlantic and 70º South 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 North or South (Reeves and Whitehead 1997). In winter, sperm whales migrate closer to equatorial waters (Kasuya and Miyashita 1988; Waring et al. 1993) 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 (2,700 nmi) apart and only rarely have been known to move over 4,000 km (2,160 nmi) 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 four km per hour. Sperm whales in the

Caribbean region appear to be much more restricted in their movements, with individuals repeatedly sighted within less than 160 km (86 nmi) of previous sightings.

4.2.7.4 Habitat

Sperm whales have a strong preference for waters deeper than 1,000 m (3,281 ft) (Reeves and Whitehead 1997; Watkins 1977)and are rarely reported from waters shallower than 300 m(Clarke 1956; Rice 1989b). However, sperm whales have been observed near Long Island, New York in water between 40 to 55 m (131 to 180 ft) deep (Scott and Sadove 1997). When observed 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 1956). 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 1996; 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. 2000; Davis et al. 2000a; Davis et al. 2000b; Davis et al. 2000c; Wormuth et al. 2000). Surface waters with sharp horizontal thermal gradients, such as along the perimeter of the Gulf Stream Current in the Atlantic, may also be temporary feeding areas for sperm whales (Griffin 1999; Jaquet et al. 1996; Waring et al. 1993). Sperm whales over George's Bank were associated with surface temperatures of 23.2 to 24.9° C (Waring et al. 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.

4.2.7.5 Reproduction

Female sperm whales become sexually mature at an average of nine years or 8.25 to 8.8 m (27.1 to 28.9 ft) (Kasuya 1991). Males reach a length of ten to 12 m (32.8 to 39.4 ft)at sexual maturity and take nine to 20 years to become sexually mature, but require another ten 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. 2004a). Adult females give birth after roughly 15 months of gestation and nurse their calves for two to three years (Waring et al. 2004a). The calving interval is estimated to be every four to six 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 1978c). 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 (six 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 six 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).

4.2.7.6 Diving and Social Behavior

Sperm whales are probably the deepest and longest diving mammalian species, with dives to 3 km (1.6 nmi) down and durations in excess of two hours (Clarke 1976; Watkins et al. 1993; Watkins et al. 1985a). However, dives are generally shorter (25 to 45 minutes) and shallower (400 to 1,000 m [1,312 to 3,281 ft]). Dives are separated by eight to 11 minutes 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 (0.3 nmi) 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 [328 to 1,640 ft]) of sperm whales in the Gulf of California overlapped with depth distributions (200 to 400 m [656 to 1,312 ft]) 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 ((greater than 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 less than 200 m (656 ft), 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 (2,700 nmi) apart and only rarely have been known to move over 4,000 km (2,160 nmi) 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 per hour. Sperm whales in the Caribbean region appear to be much more restricted in their movements, with individuals repeatedly sighted within less than 160 km (86 nmi) 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.

4.2.7.7 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 1989b). 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 1989b). The diet of large males in some areas, especially in high northern latitudes, is dominated by fish (Rice 1989b). 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).

4.2.7.8 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: 1 μPa), 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 1993; Weilgart and Whitehead 1997a). Most of the energy in sperm whale clicks is concentrated at around 2 to 4 kHz and 10 kHz to 16 kHz (Goold and Jones 1995; NMFS 2006d; Weilgart and Whitehead 1993). The highly asymmetric head anatomy of sperm whales is likely an adaptation to produce the unique clicks recorded from these animals (Cranford 1992; Norris and Harvey 1972; Norris and Harvey. 1972). Long, repeated clicks are associated with feeding and echolocation (Goold and Jones 1995; Weilgart and Whitehead 1993; Weilgart and Whitehead 1997a). However, clicks are also used in short patterns (codas) during social behavior and intragroup interactions (Weilgart and Whitehead 1993). They may also aid in intra-specific communication. Another class of sound, “squeals”, are produced with frequencies of 100 Hz to 20 kHz (e.g., Weir et al. 2007).

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

Recordings of sperm whale vocalizations reveal that they produce a variety of sounds, such as clicks, gunshots, chirps, creaks, short trumpets, pips, squeals and clangs (Goold 1999). Sperm whales typically produce short-duration repetitive broadband clicks with frequencies below 100 Hz to greater than 30 kHz (Watkins 1977)and dominant frequencies between one to six kHz and ten 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 to 60 kHz and highest sensitivity to frequencies between fiveto 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 1992a). The sperm whale may also possess better low-frequency hearing than other odontocetes, although not as low as many baleen whales (Ketten 1992a). Reactions to anthropogenic sounds can provide indirect evidence of hearing capability, and several studies have made note of changes seen in sperm whale behavior in conjunction with these sounds. For example, sperm whales have been observed to frequently stop echolocating in the presence of underwater pulses made by echo-sounders and submarine sonar (Watkins et al. 1985b; Watkins and Schevill 1975b). In the Caribbean, Watkins et al. (1985b) observed that sperm whales exposed to 3.25 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. 2007a).

A sperm whale was tagged for a controlled exposure experiment during SOCAL BRS-2010. The sperm whale did not appear to demonstrate obvious behavioral changes in dive pattern or production of clicks (Miller et al. 2012; Sivle et al. 2012; Southall et al. 2011b).

4.2.7.9 Status and Trends

Sperm whales were originally ESA-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 1995; Dohl et al. 1983; Forney et al. 1995), with peak abundance from April to mid-June and from August to mid-November (Rice 1974a).

Hill and DeMaster (1999) concluded that about 258,000 sperm whales were harvested in the North Pacific between 1947 to 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 ten 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. 2013). Little is known about the stock structure of sperm whales in the western portion of the SURTASS 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 1935b). 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 U.S. 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 809 to 9,874 m (2,670 to 32,584 ft) 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 500 m (1,640 ft) isobath; and (2) a group of six animals northwest of Saipan in waters greater than 1,000 m (3,281 ft) deep (Ligon et al. 2011). A group of ten sperm whales was also sighted during small boat surveys off western Guam in waters approximately 1,200 m (3,940 ft deep) in March 2012 (HDR 2012a).

4.2.7.10 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).

4.2.7.11 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 to 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 ten sperm whales annually (IWC 2008). In 2009, one sperm whale was killed during western North Pacific surveys (Bando et al. 2010). Whale-watching vessels are known to 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 1998c; Rice 1989a). 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 1998c). 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 approximately 32 km (17 nmi) 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. 2010a). Overall, it is estimated that 3.8 sperm whales die annually along the U.S. west coast due to fisheries interaction (Carretta et al. 2013).

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). 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).

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.

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 2009d). A fifth individual from the Pacific was found to contain nylon netting in its stomach when it washed ashore in 2004 (NMFS 2009d). 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 2009c). 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 2009c). 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).

4.2.7.12 Critical Habitat

The NMFS has not designated critical habitat for sperm whales.

4.2.8 Main Hawaiian Islands Insular False Killer Whale

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

4.2.8.1 Distribution

The MHI insular false killer whale distinct population segment occurs near the main Hawaiian Islands. Distribution of MHI insular false killer whales has been assessed using data from visual surveys and satellite tag data. Tagging data from seven groups of individuals tagged off the islands of Hawaii and Oahu indicate that the whales move rapidly and semi-regularly throughout the main Hawaiian Islands and have been documented as far as 112 km (60 nmi) offshore over a total range of 82,800 km^2 (24,141 nmi^2) (Baird et al. 2012a; Baird et al. 2012b). Three high-use areas were identified: (1) off the north half of Hawaii Island, (2) north of Maui and Molokai, and (3) southwest of Lanai (Baird et al. 2012a). However, note that limitations in the sampling, suggest the range of the population is likely underestimated, and there are probably other high-use areas that have not been identified. For example, a single satellite track suggests the potential for MHI insular false killer whales to use habitat around the Northwestern Hawaiian Islands, where a separate false killer whale DPS tends to occur (Baird et al. 2012a). Other MHI insular false killer whales tagged off of Kauai circumnavigated Niihau and returned to the northwest side of the island of Kauai.

Photo identification studies also document that the animals regularly use both leeward and windward sides of the islands (Baird et al. 2005; Baird et al. 2012a; Baird et al. 2010; Forney et al. 2010; Oleson et al. 2010). Some individual false killer whales tagged off the island of Hawaii have remained around that island for extended periods (days to weeks), but individuals from all tagged groups eventually were found broadly distributed throughout the MHIs (Baird 2009; Forney et al. 2010). Individuals utilize habitat over varying water depths less than 50 m to greater than 4,000 m (164 to 13,123 ft) (Baird et al. 2010). It has been hypothesized that interisland movements may depend on the density and movement patterns of their prey species (Baird 2009). Evidence from both tags and individual-identifying photography suggests that the area between Kauai and Niihau near the Pacific Missile Range Facility (PMRF) is an area of range overlap between two or three stocks of false killer whales, one of which is the MHI insular stock. It appears that these waters may be at the far northwestern limit of that of the MHI insular stock and the southeastern limit of the Northwestern Hawaiian Islands stock and therefore it is a question how much the animals utilize this distal area near PMRF in proportion to the rest of their larger home range (DoN 2013). Two sightings made near PMRF during U.S. Navy monitoring were not from the MHI insular stock. The first was made by U.S. Navy monitoring observers between Niihau and Kaula of an adult male, adult female, and calf, and when photographs were examined, the individuals did not match any identities in the MHI insular population catalog (R. Baird, personal communication). The second was in June 2012, when three individuals from a group of false killer whales were satellite-tagged. The tag data showed that the group subsequently left the area of PMRF, Niihau, and Kaua‘i after few days and travelled to Nihoa, Necker, French Frigate Shoals, and Gardner Pinnacles in the NWHI (Baird 2012). These two sightings comprise a sparse body of evidence that suggests the area between

Kauai and Niihau is visited by false killer whales from populations other than the MHI insular distinct population segment.

4.2.8.2 Population Structure

NMFS currently recognizes three stocks of false killer whale in Hawaiian waters under the MMPA: the Main Hawaiian Islands insular, Hawaii pelagic, and the Northwestern Hawaiian Islands stock (Bradford et al. 2012a; Carretta et al. 2011b) (77 FR 70915). NMFS considers all false killer whales found within 40 km (22 nmi) of the MHI as belonging to the Insular stock and all false killer whales beyond 140 km (76 nmi) as belonging to the Pelagic Stock (77 FR 70915). The animals belonging to the Northwest Hawaiian Islands stock are Insular to the Northwest Hawaiian Islands (Bradford et al. 2012), however, this stock was identified by animals encountered off Kauai. It has been previously recognized that the ranges for the two stocks (pelagic and insular) overlap by 100 km (54 nmi) (Bradford et al. 2012a; Carretta et al. 2011b), but there is in addition, also overlap between all three stocks given these presently identified ranges.

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

4.2.8.3 Natural Threats

Reduced genetic diversity may be a natural but partially anthropogenically induced factor leading to MHI insular false killer whale decline (Wearmouth and Sims 2008). Only a single instance of predation on false killer whales has been documented, where killer whales attacked, killed, and consumed a false killer whale calf off New Zealand (Heithaus 2001; Visser et al. 2010).

Parasitic infections have risen to levels thought to contribute to the deaths of some false killer whales, but these were from stranded individuals and it is unknown whether other health issues allowed for unhealthy levels of parasitism to develop (Andrade et al. 2001; Hernandez-Garcia 2002; Morimitsu et al. 1987; Odell et al. 1980; Sedlak-Weinstein 1991; Stacey et al. 1994; Zylber et al. 2002).

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

4.2.8.4 Anthropogenic Threats

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

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

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

4.2.8.5 Status and Trends

NMFS listed the MHI insular population of false killer whales as an endangered DPS under the ESA (77 FR 70915) in 2012. Reeves et al. (2009a) summarized information on false killer whale sightings near Hawaii between 1989 and 2007, based on various survey methods, and suggested that the MHI stock of false killer whales may have declined during the last two decades. More recently, Baird (Baird 2009) reviewed trends in sighting rates of false killer whales from aerial surveys conducted using consistent methodology around the MHIs between 1994 and 2003 (Mobley Jr 2001; Mobley Jr. 2003; Mobley Jr. 2004; Mobley Jr. 2005). Sighting rates during these surveys exhibited a statistically significant decline that could not be attributed to any weather or methodological changes. Reanalysis of previously published abundance estimates for the insular stock has led to them generally being discounted (77 FR 70915).

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

4.2.8.6 Diving and Social Behavior

Diving is not well-known in false killer whales, but individuals are believed capable of reaching 500 m (1,640 ft) in depth and possibly 700 m (2,297 ft) (Cummings and Fish 1971; Wearmouth and Sims 2008). However, most dives are significantly shallower. False killer whales occasionally dive to 150 m (492 ft) (apart from the possible 700 m dive), with frequent dives to 5 to 20 m during daytime and 30 to 40 m (98 to 131 ft) during nighttime, with durations for nighttime dives running six to seven minutes (Wearmouth and Sims 2008). Some prey, such as mahi-mahi, occur most prevalently in the top 100 m of the water column, while others, such as tuna and swordfish, may occur down to several hundred meters (Boggs 1992; Carey and Robinson 1981).

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

4.2.8.7 Vocalization and Hearing

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

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

In general, odontocetes produce sounds across the widest band of frequencies. Their social vocalizations range from a few hundreds of Hz to tens of kHz (Southall et al. 2007a) with source levels in the range of 100 to 170 dB re: 1 µPa (see (Richardson et al. 1995d). They also generate specialized clicks used in echolocation at frequencies above 100 kHz that are used to detect, localize and characterize underwater objects such as prey (Au et al. 1993). Echolocation clicks have source levels that can be as high as 229 dB re: 1 µPa peak-to-peak (Au et al. 1974).

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

4.2.8.8 Critical Habitat

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

4.2.9 Hawaiian Monk Seal

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

4.2.9.1 Distribution

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

During U.S. Navy-funded marine mammal surveys 2007 through 2012, there were 41 sightings of Hawaiian monk seals for a total of 58 individuals on (or near) Kauai, Kaula, Niihau, Oahu, and Molokai (HDR 2012b). Forty-seven (81 percent) individuals were seen during aerial surveys, and eleven (19 percent) during vessel surveys. Monk seals were most frequently observed at Niihau. Fifty-two (88 percent) individual seals were observed hauled out, and six (ten percent) were in the water as deep as 800 m (2,625 ft). In addition, seven seals were observed in August 2013, eleven in April 2013, and five in 2014 on the ledges of Kaula Island.

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

U.S. Navy-funded tagging studies in the MHIs demonstrate that mean foraging trip distance and duration, as well as maximum dive depth are similar between seals (Wilson et al. 2011). However, there were multiple outlying data points for all seals which varied by individual home ranges. Excluding one seal (R012) extended pelagic foraging trip, none of the seals traveled more than 300 km (162 nmi) per trip and most traveled less than 50 km (27 nmi) and remained within the 600 m (1,969 ft) depth contour near the MHI. The mean dive depth was 27.03 ± 44.97 m with a maximum of 529.4 m (1,737 ft) and a median depth of 14.4 m (47.2 ft). The average dive duration was 5.006 ± 3.10 minutes with a median of 5.07 minutes with 28 percent of the time between dives spent at the surface. Although foraging trip distances and durations were similar among seals, there were high levels of individual variation in where the seals travelled (Wilson and D’Amico 2012). Based on the activity budgets obtained by the tags, more accurate information is now known for modeling and estimation purposes for the percentage of time seals spend in the water and what the likely proportion of the monk seal is in the water at any given time.

4.2.9.2 Population Structure

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

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

4.2.9.3 Natural Threats

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

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

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

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

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

4.2.9.4 Anthropogenic Threats

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

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

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

4.2.9.5 Status and Trends

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

Monk seals are considered one of the most endangered groups of pinnipeds on the planet because all of their populations are either extinct (for example, the Caribbean monk seal) or exist at numbers that are precariously close to extinction (Mediterranean and Hawaiian monk seals). Two periods of decline have been reported for Hawaiian monk seals. The first decline occurred in the 1800s when sealers, crews of wrecked vessels, and guano and feather hunters nearly hunted the population to extinction (Dill and Bryan 1912; Kenyon and Rice 1959). Following the collapse of this population, expeditions to the Northwest Hawaiian Islands reported increasing numbers of seals (Bailey 1952). A survey in 1958 suggested that the population had partially recovered from its initial collapse. In June 2006, the Papahanaumokuakea Marine National Monument was established in the Northwest Hawaiian Islands (71 FR 36443). The boundary of the Monument includes about 480,187 km^2 (140,000 miles^2) of emergent and submerged lands and waters of the northwest Hawaiian Islands, and some activities such as fishing that pose risks to the marine habitat of Hawaiian monk seals are regulated.

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

(Gerrodette 1985). In 1992 the Hawaiian monk seal population was estimated to be 1,580 (standard error = 147) (Ragen 1993). The best estimate of total abundance for 1993 was 1,406 (standard error = 131, assuming a constant coefficient of variation).

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

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

The best published estimate of the total Hawaiian monk seal population size is 1,209 individuals (Carretta et al. 2014). A more recent unpublished stock assessment report estimates the total population of Hawaiian monk seals to be 1,153 individuals, with an estimated minimum population size of 1,118 individuals (Carretta et al. 2015). Population dynamics at the different locations in the northwestern Hawaiian Islands and the main Hawaiian Islands has varied considerably (Antonelis et al. 2006). The overall trend has been a steady decline, with the total number of Hawaiian monk seals decreasing from a 2007 estimate of 1,146 individuals (Littnan 2011). In the northwestern Hawaiian Islands, where most of the Hawaiian monk seals reside, the decline in abundance is approximately four percent per year. While this decline has been occurring in the northwestern Hawaiian Islands, the number of documented sightings and annual births in the main Hawaiian Islands has increased since the mid-1990s (Baker and Johanos 2004). In the MHIs, a minimum abundance of 45 seals was found in 2000, and this increased to 52 in 2001 (Baker and Johanos 2004). In 2009, 113 individual seals were identified in the main Hawaiian Islands based on flipper tag ID numbers or unique natural markings. The total number in the MHIs is estimated to be around 138 animals (Baker and Johanos 2004; Carretta et al. 2013; Carretta et al. 2015).

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

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

4.2.9.6 Diving and Social Behavior

Several recent studies of the foraging patterns of Hawaiian monk seals near rookeries in the northwestern Hawaiian Islands provide insight into their diving behavior. Monk seals spend most of their time at sea in nearshore, shallow marine habitats (Littnan et al. 2007b; Littnan and Stewart. 2007). Dive depths appear to differ slightly between rookeries as well as between age classes and genders. At Pearl and Hermes Reef, most dives were from 8 to 40 m (26 to 131 ft) with a second much smaller node at 100 to 120 m (328 to 394 ft) (Stewart 2004; Stewart et al. 2006).

At Kure Atoll, most dives were shallower than 40 m (131 ft), with males tending to dive deeper than females (Stewart and Yochem. 2004). At Laysan Island, a similar dive pattern was recorded with most dives shallower than 40 m, but at that location females tended to dive deeper than males (250 to 350 m [820 to 1,148 ft]) (Stewart and Yochem 2004). Parrish et al. (2002) noted a tendency towards night diving at French Frigate Shoals, with dives to approximately 80 to 90 m (262 to 295 ft). Based on these data, the following are rough order estimates of time at depth: 90 percent at zero to 40 m; nine percent at 40 to 120 m (131 to 394 ft); one percent at greater than 120 m. In Hawaii, overall results showed most foraging trips to last from a few days to one to two weeks, with seals remaining within the 200 m (650 ft) isobaths surrounding the MHI and nearby banks (Littnan et al. 2007a).

4.2.9.7 Vocalizations and Hearing

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

4.2.9.8 Critical habitat

Designated critical habitat for Hawaiian monk seals is discussed in Section 4.1.6.

4.2.10 Southern DPS of Spotted Seal

Spotted seals are members of the "true seal" family, Phocidae. They have a round head, narrow snout, small body, and narrow, short flippers. Their coat is silver to light gray with dark spots covering the entire body. They grow to average lengths of 1.5 m (5 ft) with weights ranging from 65 to 115 kg (140 to 250 lbs). There are few differences in size, shape, and coloration between males and females. Spotted seals are sometimes confused with harbor seals in areas where their distributions overlap, such as in Bristol Bay, Alaska.

4.2.10.1 Distribution

Spotted seals occur in cold-temperate to polar continental shelf waters of the western North Pacific Ocean from the Yangtze River to Point Barrow Alaska and as far east as the Mackenzie River Delta in the coastal waters off the North Slope of Alaska, including the Beaufort, Chukchi, southeastern East Siberian, Bering, Okhotsk, and Yellow seas, as well as the Sea of Japan (Boveng et al. 2009b; Burns and Fay 1972; Lowry 1985; Naito and Konno 1979; Naito and Nishiwaki 1972; Shaughnessy and Fay 1977). Spotted seals in the Southern DPS appear to occur year-round in two discrete breeding locations: Peter the Great Bay off Russia and Bohai Bay, especially Liaodong Bay, off China (75 FR 65239).

Spotted seals occupy a variety of habitats throughout their development as well as on a seasonal basis. Spotted seal pups are born on ice or occasionally on land and generally do not enter the water until their neonatal fur (lanugo) sheds and they grow a more insulatory adult coat (Boveng et al. 2009a). The occurrence of these pups, as well as adults, is highly dependent upon the distribution and character of seasonal ice, which provides a location for breeding, parturition, nursing, and molting away from most predators (Boveng et al. 2009a). Spotted seals generally haul-out on sea ice from late fall through spring (Bishop 1967). Sea ice tends to form earlier in more northerly latitudes and thus hauling out occurs later for more southerly individuals (Boveng et al. 2009a). In the northern Yellow Sea, spotted seals begin to occupy sea-ice habitat in Liaodong Bay from October to December, and in the western Sea of Japan they begin hauling-out in Peter the Great Bay in December (Trukhin and Mizuno 2002; Won and Yoo 2004).

Early in the season, seals select smaller ice flows (less than 20 m diameter) with thick snow for resting, parturition, nursing, weaning, and breeding (Fay 1974; Lowry 1985; Simpkins et al. 2003). Later (early- to mid-spring), spotted seals are found on such flows in pairs with an accompanying pup (Boveng et al. 2009a). Seals first haul-out where ice formation begins along shores, rivers, and estuaries and move seaward as ice continues to form (Boveng et al. 2009a; Burns 2002). Spotted seals tend to remain along the ice front (Burns 2002; Fay 1974; Heptner et al. 1976). Unlike other seals, spotted seals cannot easily scrape out a hole in the ice to breathe through, so they often must travel far over ice (greater than 10 km [5.4 nmi]) to find a naturally occurring break in the ice (Fay 1974; Fedoseev 1971). As ice melts, spotted seals become more gregarious and concentrate on the larger floes to complete molting (Quakenbush 1988).

When sea ice is not available, spotted seals may use terrestrial habitat for molting, breeding, nursing, and parturition. Some seals in Peter the Great Bay and the northern Yellow Sea have been observed to do this, particularly in low sea-ice years (Boveng et al. 2009a; Burns et al. 1981; Katin and Nesterenko 2008; Tikhomirov 1961; Tikhomirov 1964). Such terrestrial haul-outs generally consist of muddy, sandy, or gravel beaches, or rocky outcrops close to shore (Burns 2002; Heptner et al. 1976; Lowry 1985; Quakenbush 1988). Unfavorable tides, predators, or lack of food nearby are deterring factors in terrestrial haul-out selection (Heptner et al. 1976; Lowry 1985; Quakenbush 1988).

4.2.10.2 Population Structure

The total number of spotted seal populations that comprise the species is unknown. Eight breeding concentrations are known along the southern ice extent in the Pacific Ocean, but genetic and morphological distinctions between them have yet to be well described (Shaughnessy and Fay 1977). Eight breeding areas having been defined: Bohai Bay, Peter the Great Bay, Tatar Strait (western Sakhalin Island), southwest Sea of Okhotsk from eastern Sakhalin Island to Hokkaido, northeast Sea of Okhotsk (Shelikova Gulf), Karaginsky Bay (eastern side of Kamchatka Peninsula), Gulf of Anadyr (northwest Bering Sea), and eastern Bering Sea (from Bristol Bay to Pribilof Islands) (Boveng et al. 2009b; Han et al. 2010).

Based on genetic analysis, the Kamchatka Peninsula appears to prevent spotted seal populations from mixing during the breeding season; individuals from the Yellow Sea and Sea of Japan are genetically distinct from their more northerly conspecifics (Han et al. 2007; O'Corry-Crowe and Bonin 2009). Biologically, Yellow Sea and Sea of Japan spotted seals are considered a separate population, which is currently proposed for ESA-listing. All other spotted seals belong to other populations. Nakagawa et al. (2010) supported spotted seals on Hokkaido being composed of a single population based upon genetic analysis.

4.2.10.3 Movements

Spotted seals may range widely, traveling more than 1,000 km [534 nmi] round-trip in a month's time between the Kasegaluk Lagoon and the Chukchi Peninsula (Lowry et al. 1998). In general, spotted seals move into coastal waters during summer as sea ice melts and breaks up, concentrating in areas of highest prey availability (Burns 2002; Heptner et al. 1976; Lowry 1985). In the northern Yellow Sea, individuals move southward and eastward beginning in March and April.

4.2.10.4 Reproduction

Eight concentrations of spotted seal breeding areas are currently known (Shaughnessy and Fay 1977). Females give birth to a single pup annually, nurse it for two to four weeks, and then abandon it (Burns 1973; Naito and Nishiwaki 1972; Sleptsov 1943; Tikhomirov 1964; Tikhomirov 1966; Wang 1986). Pupping for seals of the Southern DPS generally occurs between early January and mid-March, with peaks between late January and late February/early March, when the sea ice is at its maximum extent and stability (Burns 2002; Burns et al. 1981; Heptner et al. 1976; Krylov et al. 1964; Tikhomirov 1964; Tikhomirov 1966; Wang 1986). Birth takes place almost exclusively on ice when available, but regularly takes place on shore for some portions of the Southern DPS (Nesterenko and Katin 2008; Nesterenko and Katin 2009; Trukhin 2005; Wang 1986). If ice melts prematurely, unweaned pups on ice are very likely to die (Burns 2002; Heptner et al. 1976). Unlike ice-born conspecifics, spotted seal pups born on land may enter the water prior to weaning (Burns 2002; Heptner et al. 1976; Katin and Nesterenko 2008; Tikhomirov 1961). Once abandoned, pups take ten to 15 days to begin feeding (Goltsev 1971).

Monogamous pairs form seven to ten days prior to parturition, and mating occurs shortly after weaning. Pairs occupying quarter to half km^2 (0.07 to 0.16 nmi^2) territories (Burns et al. 1972; Fay 1974; Heptner et al. 1976; Tikhomirov 1964). Mating likely takes place underwater and implantation of the embryo is delayed two to four months. Pregnancy lasts an average of 10.5 to 11 months (Beier and Wartzok 1979; Heptner et al. 1976).

Each year, 85 to 95 percent of mature females are reportedly pregnant (Burns 1978; Lowry 1985; Quakenbush 1988). However, infertility or abortion in females is fairly common, with rates of zero to 14 percent reported in various regions (Fedoseev 2000; Lowry 1985; Quakenbush 1988). Rates of productivity may be very high (20 to 25 percent), although it varies by region (Lowry 1985; Popov 1976; Quakenbush 1988). Mortality in early life is also high, with measured rates in excess of 40 percent in the first year, dropping to 18 percent by year two, seven percent by age three, two to four percent for ages four to six, and one to two percent for ages seven and older (Fedoseev 2000). Females six to 12 years of age have the highest fecundity (Fedoseev 2000). Adults may survive to 30 to 35 years of age, and females have a slightly longer lifespan (Naito and Nishiwaki 1972; Tikhomirov 1968).

4.2.10.5 Molting

Molting first occurs for an individual as it sheds its newborn coat, roughly two to four weeks after birth (Burns 2002; Naito and Nishiwaki 1972; Wang 1986). Once mature, molting occurs immediately after mating, when individuals gather in groups of greater than 200 individuals, likely as an anti-predatory strategy (Burns 2002; Krylov et al. 1964; Lowry 1985; Tikhomirov 1964). While molting, seals rely on fat reserves and likely have a reduced metabolic rate (Ashwell-Erickson et al. 1986). Molting for adults, particularly in the southern DPS, may be completed in water if sea ice melts (Burns 2002; Heptner et al. 1976; Tikhomirov 1964; Trukhin and Mizuno 2002).

4.2.10.6 Growth

Spotted seals are born weighing seven to 12 kg (15 to 26 lb) and are 75 to 92 cm (30 to 36 in) in length (Burns 2002; Heptner et al. 1976; Lowry 1985; Naito and Nishiwaki 1972; Tikhomirov 1968). Pups grow rapidly, increasing in weight three-fold during the two to four week nursing period (Burns 2002; Heptner et al. 1976; Tikhomirov 1968).

Adults reach sexual maturity at three to six years of age, with two-thirds of females mature by age four (Burns 1973; Fedoseev 2000; Naito and Nishiwaki 1972; Popov 1976; Sleptsov 1943; Tikhomirov 1966). Age at sexual maturity varies based upon several factors, including selectivity and favorability of environmental conditions and hunting intensity (Quakenbush et al. 2009). Females stop growing at roughly ten years of age, while males may grow for another five years, although growth in both sexes appears to slow significantly at eight or nine years of age (Naito and Nishiwaki 1972; Tikhomirov 1968). Adults appear to generally reach maximum sizes of 162 cm (64 in) for females and 168 cm (66 in) for males, although a 182 cm (72 in) female and a 185 cm (73 in) male have been reported (Tikhomirov 1968). The spotted seal sex ratio appears to be even through age nine to ten, but females may compose a higher proportion of older individuals (Fedoseev 2000; Lowry 1985; Sleptsov 1943).

4.2.10.7 Feeding

The diet of spotted seals varies based upon seal maturity. Pups consume primarily or exclusively amphipods, euphausiids, and other crustaceans, but may also eat Pacific sand lance (Bukhtiyarov 1990; Bukhtiyarov et al. 1984; Goltsev 1971; Kato 1982). Juveniles shift to a more fish-based diet, although they may continue eating euphausiids (Bukhtiyarov et al. 1984; Kato 1982; Sobolevsky 1996). Subadults consume larger quantities of fish, large shrimp, and fewer crustaceans.

As adults, fish compose almost the entire diet and some cephalopods are consumed at all ages (Bukhtiyarov 1990; Bukhtiyarov et al. 1984; Goltsev 1971; Kato 1982; Sobolevsky 1996). This dietary pattern may not hold steady over decadal periods, as younger individuals during the 2000s seemed to eat fish as frequently as older individuals (Quakenbush et al. 2009). Invertebrate prey have recently been found to be a smaller component of spotted seal diet than in the 1960s and 1970s (Quakenbush et al. 2009).

Diet may also vary further based upon the regional and seasonal prey distribution, although little information is available from individuals known to be from the southern DPS (Bukhtiyarov et al. 1984). During spring in the Bering Sea, Sea of Okhotsk, and Chukchi Sea, seals associating with ice consume walleye pollock, Pacific herring, Arctic cod, Pacific sand lance, capelin, saffron cod, Japanese smelt, as well as greenlings, Okhotsk mackerel, eelpouts, sculpins, flatfishes, cephalopods, and crustaceans (Bukhtiyarov 1990; Bukhtiyarov et al. 1984; Dehn et al. 2007; Fedoseev and Bukhtiyarov 1972; Frost et al. 1977; Goltsev 1971; Huntington 2000; Kato 1982; Lowry 1985; Lowry et al. 1977; Lowry et al. 1978; Lowry et al. 1979a; Lowry et al. 1981; Lowry et al. 1979b; Nikolaev and Skalkin 1975; Sobolevsky 1996; Wilke 1954). Summer and early fall diet is similar, but may concentrate more on salmon runs in coastal streams and rivers, where pink and coho salmon and Arctic char may compose 80 percent of a seal’s diet (Burkanov 1989; Huntington 2000; Kosygin et al. 1986; Sobolevsky 1996). Fall and winter diet is less well known, but is believed to include herring, walleye pollock, Pacific sand lance, capelin, smelt, saffron cod, Arctic cod, octopuses, crabs, and shrimp (Huntington 2000; Lowry 1985; Sobolevsky 1996), although mollusks and worms may also be important during winter (Barabash-Nikiforov 1938). Spotted seals appear to consume pollock of a smaller size than what is generally harvested by commercial fisheries (Frost and Lowry 1986; Ianelli et al. 2008).

4.2.10.8 Diving and Social Behavior

Spotted seals are unique among seals in several behavioral respects. Mating is monogamous in a given season and females cannot identify their own pups (Burns 2002; Burns et al. 1972; Quakenbush 1988; Tikhomirov 1964). Spotted seals are very wary and are easily disturbed from their haul-out sites and, if frequently disturbed, females will abandon their pups (Frost et al. 1993; Heptner et al. 1976; Krylov et al. 1964; Lowry 1985; Tikhomirov 1964; Wang 1986).

Spotted seal diving ability increases with maturity, with pups capable of 80 m (262 ft) dives, and adults likely reaching depths of 300 m (984 ft) (Goltsev 1971). However, as habitat is primarily in waters over the continental shelf, most dives are less than 200 m (656 ft) deep (Bukhtiyarov et al. 1984; Lowry 1985). Swimming speeds of up to 3.8 m per second have been recorded, and seals are known to be able to travel at least 400 m (1,312 ft) horizontally underwater on a single breath (Chugunkov 1970).

4.2.10.9 Vocalization and Hearing

Little is known about the hearing of spotted seals. Recently, Sills et al. (2014) measured the underwater and in-air hearing capabilities of young spotted seals and found that the best hearing sensitivity in air spanned four octaves, ranging from approximately 0.6 to 11 kHz, while the hearing sensitivity underwater ranged from 0.3 and 56 kHz. Sills et al. (2014) concluded that the sound reception capabilities of spotted seals differ from those described previously for ice seals, with capabilities more similar to harbor seals. Five distinct sounds have been identified in captive spotted seals of both sexes: growls, drums, snorts, chirps, and barks that range in frequency from 500 Hz to 3.5 kHz (Beier and Wartzok 1979; Richardson et al. 1995a). A “creaky door sound” has only been recorded from males (Beier and Wartzok 1979).

4.2.10.10 Status and Trends

In October 2010, NMFS listed the Southern DPS of spotted seals, which consists of breeding concentrations in the Yellow Sea (particularly the Bohai Sea, a northwestern branch of the Yellow Sea, China) and Peter the Great Bay (northwestern Sea of Japan, Russia), as threatened under the ESA (75 FR 65239).

Spotted seal abundances, where estimated, are likely very inaccurate due to the species’ skittish nature and unknown proportion of individuals in-water versus on ice/land at time of survey (Taylor et al. 2007). Several abundance estimates exist for portions of the Southern DPS. Liaodong Bay was estimated to have approximately 7,100 individuals in 1930 and a maximum of 8,137 in 1940. The most recent abundance estimate, in 2007, for the Liaodong Bay population is around 800 individuals. Peter the Great Bay likely consisted of several thousand individuals historically, but declined to several hundred due to hunting pressure by 1968. Over 1,000 individuals were present by 1978, and the abundance remained stable through at least 1996 (Trukhin and Mizuno 2002). The most recent estimate for Peter the Great Bay, the population that overlaps with the Sea of Japan mission areas within the SURTASS action area, is approximately 2,500 individuals, with production of approximately 300 pups annually on shore and not on ice (Nesterenko and Katin 2008; Nesterenko and Katin 2009). Between the Liaodon Bay and Peter the Great Bay populations, the Southern DPS of spotted seals has an estimated 3,300 individuals.

4.2.10.11 Natural Threats

Spotted seal predators include polar bears, brown bears, walruses, killer whales, Pacific sleeper sharks, foxes, wolves, sea lions, eagles, and gulls (Popov 1982; Quakenbush 1988). Brown bears were reportedly the primary predator of spotted seals in the Sea of Okhotsk (Popov 1976).

4.2.10.12 Anthropogenic Threats

Spotted seals have long been a target of commercial and subsistence hunting, but threats from climate change and environmental contaminants are more recent developments, which prompted the Southern DPS to be proposed for ESA-listing. Spotted seals are still hunted for meat, fur, oil, and animal feed by coastal inhabitants of the northern Pacific Rim (Fedoseev 1984; Krylov et al. 1964).

Before 1950, subsistence hunting in the Sea of Okhotsk averaged 3,750 to 5,250 seals per year, with additional unknown harvest levels from the Bering and Chukchi coasts (Fedoseev 1984; Fedoseev 2000; Krylov et al. 1964). Harvests in Peter the Great Bay were roughly 80 individuals annually until approximately 1900, when catches began to increase (Trukhin and Mizuno 2002). Spotted seals are treated as a threat to fisheries and are bycaught or shot, which is still considered the greatest threat to spotted seals in Peter the Great Bay (Trukhin and Mizuno 2002). During the winter of 1996 to 1997, it is estimated that 100 to 150 spotted seals were killed in nets. This mortality level is considered unsustainable and likely limited spotted seal population growth in the Peter the Great Bay region during the 1990’s (Trukhin and Mizuno 2002).

From 1971 to 1998, 113 bycaught and 153 “damage control kills” occurred around Hokkaido, Japan (Trukhin and Mizuno 2002). Seals in this region were also collected for museums, zoos, and aquaria (Mizuno et al. 2001). Hunting in the northern Yellow and Bohai seas is considered to be the reason for spotted seal declines here, with an average of 500 and up to 1,000 individuals harvested annually from 1930 to 1990 (Dong and Shen 1991; Wang 1998). In most or all of these areas, subsistence harvesting, bycatch, and “control kills” likely continue today.

Subsistence harvest in Alaska has amounted to several thousand individuals killed annually, with reported levels of approximately 2,400 individuals harvested from 1966 to 1976, and 986 from 1985 to 1986 between five native villages, and 210 from six native villages between 1992 to 2006 (Lowry 1985; Quakenbush 1988; Wolfe et al. 2008). Overall, the Alaska Department of Fish and Game estimates that 5,265 spotted seals are harvested annually (Angliss and Allen 2009).

Commercial harvests continue to be conducted for spotted seals along Asian coasts and territories. Historically, 500 to 1,000 spotted seals were annually killed from western Sea of Okhotsk and Tatar Strait before Russian-sponsored sealing at rookeries ended in the late 1930s (Heptner et al. 1976). Okhotsk sealing from ships grew in the mid-1950s as fleets grew, with annual kills averaging 4,634 (and likely substantially larger due to unreported kills) until the industry became regulated in 1969 (Fedoseev 1984; Fedoseev 2000; Heptner et al. 1976; Krylov et al. 1964). Similarly, Bering Sea kills swelled to approximately 4,600 annually, (adjusted for underreporting; (Fedoseev 2000). Regulation did little to curb harvesting. According to some estimates, kills in the Sea of Okhotsk and Bering Sea combining to 10,000 to 15,000 individuals annually up to the mid-1970s (Popov 1976); however Lowry (1985) reported that the total annual Soviet harvest in the Bering and Chukchi Seas during 1966 to 1976 averaged about 3,850.

Harvests increased from the early 1970s due to the fur trade, with roughly 4,800 individuals from the Bering Sea and approximately 4,300 from the Sea of Okhotsk killed annually through 1985 and 1994, respectively (Fedoseev 2000; Grachev 2006). However, the Sea of Okhotsk harvest dropped to roughly 400 individuals from 1995 to 2005 (Grachev 2006). A few hundred individuals were also harvested annually from the Chukchi Sea (Quakenbush 1988). Russian authorities have proposed new catch limits for spotted seals as 4,500 individuals annually, but it is not known if this limit has or will be set (Grachev 2006). Approximately 1,100 spotted seals were harvested annually from Hokkaido during 1968-1970 (Naito 1971).

Climate change is also a concern for spotted seals. Although significant direct effects are not expected, indirect effects can be important (Boily 1995; Grebmeier et al. 2006; Harding et al. 2005). These include changes in prey abundance or distribution, predation, and disease (Boveng et al. 2009a; Comeau et al. 2009; Grebmeier et al. 2006).

Ocean contaminants are also potential threats. Liver tissues have been found to contain cadmium, mercury, and other metals, although these levels are generally considered to be low (Dehn et al. 2005; Quakenbush et al. 2009). An exception is methyl mercury, which is higher in spotted seals than in other ice seals, likely due to the fish-eating diet of adults (Dehn et al. 2005; Quakenbush et al. 2009). A variety of organic contaminants, including bioaccumulative and toxic (BTs), persistant organic pollutants (POPs), polybrominated diphenyl ethers (PBDEs), DDT, PCBs, perfluorochemicals (PFCs), chloroform (CHL), and HCH, have been isolated from spotted seal tissues in generally low levels (with the exceptions of PCB, DDT, and HCH), which have been linked to immune dysfunction, reproductive failure and disorder, as well as thyroid function in pinnipeds (Chiba et al. 2001; Chiba et al. 2002; de Swart et al. 1996; Gregory and Cyr 2003; Iwata et al. 1997; Nakata et al. 2002; Neale et al. 2007; Quakenbush et al. 2009; Quakenbush 2007; Quakenbush and Citta 2008; Ross et al. 1996; Tanabe et al. 1998).

4.2.10.13 Critical Habitat

Critical habitat has not been designated for the Southern DPS of the spotted seal.

4.2.11 Steller Sea Lion – Western DPS

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

4.2.11.1 Distribution

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

4.2.11.2 Reproduction

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

Mothers with newborn pups will make their first foraging trip about a week after giving birth, but trips are short in duration and distance at first, then increase as the pup gets older (Maniscalco et al. 2006; Merrick and Loughlin 1997; Milette 1999; Milette and Trites 2003; Pitcher et al. 2001). Females attending pups tend to stay within 37 km (20 nmi) of the rookery (Calkins 1996; Merrick and Loughlin 1997). Newborn pups are wholly dependent upon their mother for milk during at least their first three months of life, and observations suggest they continue to be highly dependent upon their mother through their first winter (Porter 1997; Scheffer 1945; Trites et al. 2006). Generally, female Steller sea lion will nurse their offspring until they are one to two years old (Calkins and Pitcher 1982b; Gentry 1970; Pitcher and Calkins 1981; Sandegren 1970; Trites et al. 2006). Pups may enter the water after two to four weeks (Sandegren 1970).

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

4.2.11.3 Habitat

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

Adult females may travel far out to sea into water greater than 1,000 m deep (Merrick and Loughlin 1997). Studies on immature Steller sea lions indicate three types of movements: long-range trips (greater than 15 km [8 nmi] and greater than 20 hours), short-range trips (less than 15 km and less than 20 hours), and transits to other sites (NMFS 2007a). Long-range trips started around nine months of age and likely occur most frequently around the time of weaning, while short-range trips happen almost daily. Young individuals generally remain within 480 km (259 nmi) of rookeries their first year before moving further away in subsequent years (Raum-Suryan et al. 2004). Many animals also use traditional rafting sites, which are places where they rest on the ocean surface in a tightly packed group (Bigg 1985). Frontal features with small-scale temperature gradients appear to be attractive foraging sites for juvenile Steller sea lions (Lander et al. 2010). Large numbers of Steller sea lions are found near the 200 m isobath year round (Consiglieri et al. 1982). Foraging generally occurs within 8 to 24 km (4.3 to 13 nmi) of shore (Fiscus and Braham 1976). However, foraging can occur hundreds of kilometers from shore over extended periods (Merrick et al. 1997).

4.2.11.4 Feeding

Steller sea lions are generalist predators that eat various fish (arrowtooth flounder, rockfish, hake, flatfish, Pacific salmon, Pacific herring, Pacific cod, sand lance, skates, cusk eel, lamprey, walleye, Atka mackerel), squids, octopus, and occasionally birds and marine mammals (Brown et al. 2002; Calkins and Goodwin 1988; Daniel and Schneeweis 1992; Jones 1981; McKenzie and Wynne 2008; Olesiuk et al. 1990; Pitcher and Fay 1982; Sinclair and Zeppelin 2002; Womble and Conlon. 2010). Diet is likely strongly influenced by local and temporal changes in prey distribution and abundance (McKenzie and Wynne 2008; Sigler et al. 2009). Haulout selection appears to be driven at least in part by local prey density (Winter et al. 2009). Adult females embark on foraging trips of at night for 7 to 26 hours during the breeding season, while adult males rarely or never eat while on breeding grounds (Andrews et al. 2001; Loughlin 2002a).

4.2.11.5 Diving

Diving activity is highly variable in Steller sea lion by sex and season. During the breeding season, when both males and females occupy rookeries, adult breeding males rarely, if ever, leave the beach (Loughlin 2002b). However, females tend to feed at night on one to two day trips and return to nurse pups (NRC 2003a). Female foraging trips during winter are longer (130 km [70 nmi]) and dives are deeper (frequently greater than 250 m [820 ft]). Summer foraging dives, however, are closer to shore (about 16 km [8.6 nmi]) and shallower (100 to 250 m [328 to 820 ft]; (Loughlin 2002b; Merrick and Loughlin 1997)). As pups mature and start foraging for themselves, they develop greater diving ability until roughly ten years of age (Pitcher et al. 2005). Juveniles usually make shallow dives of 70 to 140 m (230 to 459 ft) over one to two minutes, but much deeper dives in excess of 300 m (984 ft) are known (Loughlin et al. 2003; Merrick and Loughlin 1997; Rehberg et al. 2001). Young animals also tend to stay in shallower water less than 100 m deep and within 20 km (10.8 nmi) from shore (Fadely et al. 2005).

4.2.11.6 Acoustics and Hearing

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

4.2.11.7 Status and Trends

Steller sea lions were originally listed as threatened under the ESA on November 26, 1990 (55 FR 49204), following a decline in the U.S. of about 64 percent over the previous three decades. In 1997, the species was split into two separate populations based on demographic and genetic differences (Bickham et al. 1996; Loughlin 1997), and the western population was reclassified to endangered (62 FR 24345) while the eastern population remained threatened (62 FR 30772). On April 18, 2012, the NMFS proposed to delist the eastern DPS of Steller sea lions (77 FR 23209). On November 4, 2013, the NMFS announced that as of December 4, 2013, the eastern DPS of Steller sea lions would be delisted and no longer protected under the ESA (78 FR 66139). The 2013-2014 Biennial Report to Congress states the status of the Western DPS of this species is considered mixed. The Steller sea lion is listed as near threatened on the 2012 IUCN Red List.

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

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

4.2.11.8 Natural Threats

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

Steller sea lions have tested positive for several pathogens, but disease levels are unknown (FOC 2008). Similarly, parasites in this species are common, but mortality resulting from infestation is unknown. However, significant negative effects of these factors may occur in combination with stress, which reduces immune capability to resist infections and infestations. If other factors, such as disturbance, injury, or difficulty feeding occur, disease and parasitism may play a greater role in population reduction.

4.2.11.9 Anthropogenic Threats

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

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

Contaminants are a considerable issue for Steller sea lions. Roughly 30 individuals died as a result of the Exxon *Valdez* oil spill and contained particularly high levels of polycyclic aromatic hydrocarbons (PAH) contaminants, presumably as a result of the spill. Blood testing confirmed hydrocarbon exposure. Subsequently, premature birth rates increased and pup survival decreased (Calkins et al. 1994; Loughlin et al. 1996). Organochlorines, including PCBs and DDT (and their metabolites), have been identified in Steller sea lions in greater concentrations than any other pinniped during the 1980s, although levels appear to be declining (Barron et al. 2003; Hoshino et al. 2006). The levels of PCBs have been found to have twice the burden in individuals from Russia than from western Alaska (4.3 ng/g wet weight versus 2.1 ng/g wet weight; (Myers et al. 2008). Levels of DDT in Russian pups were also on average twice that in western Alaska pups (3.3 ng/g wet weight blood versus 1.6 ng/g wet weight). PCB levels in the kidneys of some adult males are high enough that reproductive and immune function may have been compromised (Wang et al. 2011a). The source of contamination is likely from pollock, which have been found to contain organochlorines throughout the Gulf of Alaska, but higher in regions occupied by the eastern DPS of Steller sea lions (NMFS 2008c). Heavy metals, including mercury, zinc, copper, metallothionien, and butyltin have been identified in Steller sea lion tissues, but are in concentrations lower than other pinnipeds (Beckmen et al. 2002; Castellini 1999; Kim et al. 1996; NMFS 2008c; Noda et al. 1995). Mercury may be of higher significance, with liver levels being measured at levels above those necessary to impact fish (Holmes et al. 2008). However, contaminants leading to mortality in Steller sea lions have not been identified (NMFS 2008c). Contaminant burdens are lower in females than males, because contaminants are transferred to the fetus *in utero* as well as through lactation (Lee et al. 1996; Myers et al. 2008). However, this means that new generations tend to start with higher levels of contaminants than their parents originally had. Steller sea lion contaminants are of additional concern because contaminants in the body tend to be mobilized as fat reserves are used, such as when prey availability is low; a situation that is likely occurring for Steller sea lions today.

4.2.12 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 5 cm (2 in) long, adults can grow to more than 0.91 m (3 ft) long and weigh 136 to 159 kg (300 to 350 lb).

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.

4.2.12.1 Populations

The population dynamics of green turtles and all of the other sea turtles we consider in this opinion and conference report 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 1998d; 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.

4.2.12.2 Distribution

Green turtles are found in the Pacific Ocean, Atlantic Ocean, Indian Ocean, Caribbean 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 Caribbean 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).

4.2.12.3 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 (7.9 to 9.8 in) carapace length, juveniles leave pelagic habitats and enter benthic foraging areas (Bjorndal 1997). 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 (32.8 ft) when moderate depths of 5 to 10 m (16.4 to 32.8 ft) 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). East Pacific adults migrate along coastal corridors between Central American nesting and foraging locations (Blanco et al. 2012).

4.2.12.4 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 episodes. 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 1998c). 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).

4.2.12.5 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 less than one cm (0.4 in) per year (Green 1993) to greater than five cm (two in) per 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 (0 to 3.7 in) annually (mean of 4.1, SD 2.4). The largest growth rates were in the 30 to 39 cm (11.8 to 3.5 in) 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 range from approximately 20 to 40 years or more (Balazs 1982; Chaloupka et al. 2004b; Chaloupka and Musick 1997; Frazer and Ehrhart 1985a; 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 durations between females returning to nest range from two to five years (Hirth 1997), these reproductive longevity estimates suggest that a female may nest three to 11 seasons over the course of her life. Each female deposits one to seven clutches (usually two to three) during the breeding season at 12 to 14 day intervals. Mean clutch size is highly variable among populations, but averages 110 to 115 eggs per nest. Females usually have two to four 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 nine 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 after the first few hours (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 in the foraging area, green turtles 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 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).

4.2.12.6 Feeding

Green sea turtles are largely herbivorous as juveniles and adults, but are omnivorous in earlier lifestages. 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, and sea pens (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 plant matter apparently replaces an omnivorous diet off Mauritania when green turtle’s carapace is around 59 cm (23.2 in) in length (Cardona et al. 2009). This transition may occur rapidly starting at 30 cm (11.8 in) carapace length, but animal prey continue to constitute an important nutritional component until individuals reach about 62 cm (24.4 in) (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 green turtle populations can consume a variety of prey and are considered generalists as a whole, individuals maintain a highly-selective diet over long time frames (Vander Zanden et al. 2013).

4.2.12.7 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 1998c). Recent data from Australia indicate green sea turtles rarely dive deep, staying in the upper eight meters 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 dove to a maximum depth greater than (135.6 m (445 ft) at night (the deepest dives ever recorded for a green turtle) with a mean maximum night dive depth of 35 to 50 m (115 to 164 ft), while only diving to depths of 4.3 m (14.1 ft) during the day (Rice and Balazs 2008b). In their coastal habitat, green turtles typically make dives shallower than 30.5 m (100 ft) (Hatase et al. 2006a; Hays et al. 2000; Hochscheid et al. 2005; Houghton et al. 2002) and often do not exceed 16.8 m (55 ft) (Hays et al. 2000; Rice and Balazs 2008a), although they are known to feed and rest at depths of 19.8 to 50.3 m (65 to 165 ft) (Balazs 1980; Brill et al. 1995). Also, time spent resting and dive durations increased significantly as water temperatures seasonally decreased. Subadults routinely dive to 20 m for nine to 23 minutes, with a maximum recorded dive of over one hour (Brill et al. 1995; 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).

4.2.12.8 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. 1999b; Lenhardt 1994; 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 1994). 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. 1999b). However, Dow et al. (2008) found best sensitivity between 50 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).

4.2.12.9 Natural Threats

Herons, gulls, dogfish, and sharks prey upon hatchlings. Predators (primarily of eggs and hatchlings) also include dogs, pigs, rats, crabs, sea birds, reef fishes, and groupers (Bell et al. 1994; Witzell 1981). Adult green 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. For unknown reasons, the frequency of a disease called fibropapillomatosis is much higher in green sea turtles than in other species and threatens a large number of existing subpopulations. Extremely high incidences of fibropapillomatosis have been reported in Hawaii, where affliction rates peaked at 47–69 percent in some foraging areas (Murakawa et al. 2000). 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).

4.2.12.10 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, greater than 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 1991). 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. 2006; 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).

4.2.12.11 Green Sea Turtle DPSs

On April 6, 2016, NMFS published a final rule to list 11 DPSs of green sea turtles globally as threatened or endangered under the ESA (Figure 7; 81 FR 20057). Eight DPSs are ESA-listed as threatened: Central North Pacific, East Indian-West Pacific, East Pacific, North Atlantic, North Indian, South Atlantic, Southwest Indian, and Southwest Pacific. Three DPSs are ESA-listed as endangered: Central South Pacific, Central West Pacific, and Mediterranean. Individuals from the Central North Pacific, Central West Pacific, East Indian-West Pacific, and North Indian DPSs are likely to be affected by the proposed action.

Green sea turtle DPSs were delineated according to nesting beaches. The DPS delineation map below (Figure 7) 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 Exclusive Economic Zone (EEZ) primarily around the MHIs 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 2015). This indicates the oceanic range of green sea turtle DPSs extends well beyond the DPS delineations presented in Figure 7.

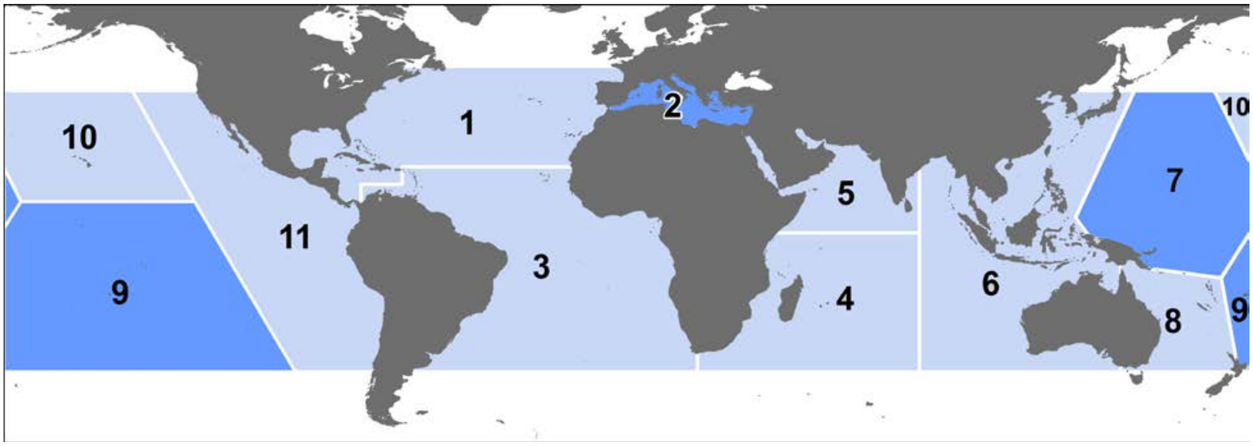

Figure 7. Map depicting ESA-listed DPS boundaries for green sea turtles. Threatened (light blue) and endangered (dark blue) green turtle DPSs: 1) North Atlantic, 2) Mediterranean, 3) South Atlantic, 4) Southwest Indian, 5) North Indian, 6) East Indian-West Pacific, 7) Central West Pacific, 8) Southwest Pacific, 9) Central South Pacific, 10) Central North Pacific, and 11) East Pacific (81 FR 20057).

Beyond DPS designations, populations are distinguished generally by ocean basin and more specifically by nesting location. Worldwide, nesting data at 46 sites from 1990 to 2006 indicate that 108,761 to 150,521 females nest each year. Nesting populations are doing relatively well in the Pacific, Western Atlantic and Central Atlantic Ocean; whereas, populations are doing poorly in Southeast Asia, Eastern Indian Ocean and Mediterranean.

Table 17. Green sea turtle nesting abundance for DPSs within the action area. Adapted from Seminoff et al. 2015.

Green Sea Turtle DPS	Central West Pacific	Central North Pacific	East Indian-West Pacific	North Indian
ESA Status	Endangered	Threatened	Threatened	Threatened
Total Number Nesting Sites	50	13	50	38
Total Nester Abundance	6,518	3,846	77,009	55,243
Population Trend (PVA)	Unknown	Increasing	Unknown	Unknown

The green sea turtles in the action area would occur in the Central North Pacific DPS (mission areas 10 and 11), the endangered Central West Pacific DPS (mission areas 1, 4, 8, and 9), the threatened East Indian-West Pacific DPS (mission areas 2, 3, 5, 6, 7, 13, and 14), and the threatened North Indian DPS (mission area 12). The distribution range of green sea turtles does not extend as far north as the waters of mission area 15.

4.2.12.12 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, 1980; Lipman and Balazs, 1983). Current nesting by green turtles occurs in low numbers (3 to 36 nesting females at any one site) throughout the Northwest Hawaiian Islands 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 to 24) on seven islands (Frey *et al.,* 2013; Kittinger *et al.,* 2013; NMFS Pacific Islands Fisheries Science Center, unpublished data, 2013). Green turtles in the Central North Pacific DPS bask on beaches throughout the Northwest Hawaiian Islands 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, 2006). During the first five years of monitoring (1973 to 1977), the mean annual nesting abundance was 83 females, and during the most recent five years of monitoring (2009 to 2012), the mean annual nesting abundance was 464 females (Balazs and Chaloupka, 2006; G. Balazs, NMFS, 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 (Balazs, 2000; Balazs *et al.*, 2005; Balazs *et al.,* 1996). 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 two 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 around 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 above, 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.

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 (Ogasawara Islands). 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 50 nesting sites and 6,518 nesting females in the Central West Pacific DPS. Unquantified nesting sites, possibly with small numbers, exist, but 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) (NMFS and FWS, 1998; Bureau of Marine Resources 2005; Barr 2006; Palau Bureau of Marine Resources 2008; Maison et al., 2010; NMFS and FWS 2015).

There are numerous other populations in the FSM, Solomon Islands, and Palau, and approximately 22 nesting green sea turtles in Guam, and 57 nesting green sea turtles in 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 (Groombridge and Luxmoore 1989).

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 (594 nmi) away (Klain et al., 2007 in NMFS & FWS 2015). 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 (Kabua et al., 2012 in NMFS & FWS 2015). Sea turtles tagged in Yap (FSM) were recaptured in the Philippines, Marshall Islands, Papua New Guinea, Palau, and Yap (Palau BMR 2008; Cruce 2009). A sea 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 (Palau BMR, 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 & FWS 2015). A turtle tagged in Japan was recorded nesting in Yap, FSM (Cruce 2009 in NMFS & FWS 2015).

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 (DAWR 2011 in NMFS & FWS 2015). In-water green turtle density in the Marianas Archipelago is low and mostly restricted to juveniles (Pultz et al., 1999; Kolinski et al., 2005, 2006; Palacios 2012). 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. 2010a; Maison et al. 2010b).

In addition to the natural and anthropogenic threats to all DPSs of green sea turtles, as described above, 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) and final rule (81 FR 20058). 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.

4.2.12.13 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 East Indian-West Pacific DPS exhibits high abundance with an estimated total nester abundance of 77,009 females at 50 nesting sites. The largest nesting site (Wellesley Group in northern Australia) supports approximately 25,000 nesting females. Green sea turtles within this DPS have experienced increases at some nesting sites and decreases at others. Nonetheless, abundance are substantially depleted from historical levels.

There is complex and significant spatial substructure, but some mixing of sea turtles occurs at foraging areas. 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 2015). 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 above, threats specific to the East Indian-West Pacific DPS are discussed in the proposed rule (80 FR 15271) and final rule (81 FR 20058). 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.

4.2.12.14 North Indian DPS

The nesting range of the North Indian DPS begins at the border of Somalia and Kenya north into the Gulf of Aden, Red Sea, Persian Gulf, and east to the Gulf of Mannar off the southern tip of India and includes a major portion of Indian’s southeastern coast up to Andra Pradesh. The southern and eastern boundaries are the equator and 84º East, respectively, which intersect in the southeastern corner of the range of the DPS. It is bordered by the following countries (following the water bodies from west to east): Somalia, Djibouti, Eritrea, Sudan, Egypt, Israel, Jordan, Saudi Arabia, Yemen, Oman, United Arab Emirates, Qatar, Bahrain, Kuwait, Ira, the Islamic Republic of Iran, Pakistan, India, and Sri Lanka.

The North Indian DPS exhibits high abundance, with an estimated total nester abundance of 55,243 females at 38 nesting sites. Two sites host greater than 10,000 nestig females: Ras Sharma, Yemen, and Ras Al Hadd, Oman. Nesting trends are increasing at Ras Al Hadd, but possibly declining at other sites. Nesting is moderately dispersed, though concentrate din the northern and western region of the range.

In addition to the natural and anthropogenic threats to all DPSs of green sea turtles, as described above, threats specific to the North Indian DPS are discussed in the proposed rule (80 FR 15271) and final rule (80 FR 20058). Threats include, but are not limited to, direct harvest, incidental bycatch in fisheries, destruction and modification of nesting habitat, disease, predation, marine debris, beach driving, vessel strikes, and climate change.

4.2.12.15 Critical Habitat

On September 2, 1998, critical habitat for green sea turtles was designated in coastal waters surrounding Culebra Island, Puerto Rico (63 FR 46693). Aspects of these areas that are important for green sea turtle survival and recovery include important natal development habitat, refuge from predation, shelter between foraging periods, and food for green sea turtle prey. Since green sea turtle designated critical habitat does not occur in the action area for SURTASS LFA sonar, it will not be addressed further in this opinion and conference report.

4.2.13 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.

4.2.13.1 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.

4.2.13.2 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 occur at weed-lines that accumulate at convergence points. When they grow to about 20 to 25 cm (7.8 to 9.8 in) in 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 1,601 km (995 miles) 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.

4.2.13.3 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 (154 nmi), eventually taking up residency in the deeper waters outside of Cocos Lagoon, Guam (Jones and Houtan 2014).

4.2.13.4 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 [2 to 8.3 in] 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 beds, 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 2007d). 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 from 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).

4.2.13.5 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 three to five 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 (8.7 to 9.8 in) 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 (25.6 to 27.6 in) in curved carapace length, after which it slows to negligible amounts after 80 cm (31.5 in) (Campbell et al. 2012). As with other sea turtles, growth is variable and likely depends upon nutrition available (Campbell et al. 2012). Juvenile hawksbills along the British Virgin Islands grow at a relatively rapid rate of roughly 9.3 cm (3.7 in) per year and gain 3.9 kg (8.6 lb) annually (Hawkes et al. 2014).

4.2.13.6 Feeding

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

4.2.13.7 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 minutes) compared to daytime (19 to 26 minutes) (Blumenthal et al. 2009b; Van Dam and Diez 1997). Daytime dives averaged 5 m (16.4 ft), while nighttime dives averaged 43 m (141.1 ft) (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 minutes, with a maximum time of 73.5 minutes (Starbird et al. 1999). Average day and night dive times were 34 to 65 and 42 to

74 minutes, respectively. Immature individuals have much shorter dives of 8.6 to 14 minutes to a mean depth of 4.7 m (15.4 ft) while foraging (Van Dam and Diez 1997).

4.2.13.8 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. 1999b; Lenhardt 1994; 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).

4.2.13.9 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 2007d). 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 five to ten females are estimated to nest annually (G. Balazs, NMFS, in litt. to J. Mortimer 2007; G. Davis, NMFS, in litt. to J. Mortimer 2007) and the same is true for Hawaii, but there are indications that this population is increasing (G. Balazs, pers. comm. in NMFS and USFWS 2007d). 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 = 5,430) 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). Total number of nesting females for the Central Pacific hawksbill population was estimated at 940 to 1,200 females annually for the last few years, with an overall downward trend (NMFS and USFWS 2007b). 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).

4.2.13.10 Natural Threats

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), sharks, and to a lesser extent, killer whales also prey on hawksbills. 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).

4.2.13.11 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 1991).

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 to comply with Convention on International Trade in Endangered Species of Wild Flora and Fauna (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 1993b). 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 1993b). 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).

4.2.13.12 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. Since hawksbill sea turtle critical habitat does not occur in the action area for SURTASS LFA sonar, it will not be addressed further in this opinion and conference report.

4.2.14 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 92 cm (36 in); corresponding weight is about 113 kg (250 lb).

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 4.6 cm (1.8 in) in length and weigh approximately 20 grams (g) (0.04 lb).

4.2.14.1 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 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. 1995; Resendiz et al. 1998). 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).

4.2.14.2 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 on 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, the Caribbean, and into South America. Within the Indian Ocean, nesting aggregations occur at Oman, Yemen, Sri Lanka, 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); 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 and 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.

4.2.14.3 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 2001a). Adult loggerheads make lengthy migrations from nesting beaches to foraging grounds (TEWG 1998b). 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).

During aerial surveys sponsored by the U.S. Navy from January to August 2009, 193 loggerhead turtles were sighted 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 in the same area 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 in the same area 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 Current) to feed for several years before migrating back to western Pacific waters to breed (Bowen et al. 1995; Nichols 2005; Polovina et al. 2006; Polovina et al. 2000; Resendiz et al. 1998). 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 15to 25º C, or 28to 40º North, 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. 2000b; TEWG 1998a). 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).

4.2.14.4 Growth and Reproduction

Loggerhead nesting is confined to lower latitudes temperate and subtropic zones but is 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 seven to 12 years (NMFS 2005a). 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 per year for the first six months and slow to roughly 3.6 cm (1.4 in) per year 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 1985b; Frazer et al. 1994; NMFS 2001a; Witherington et al. 2006b). However, based on new data from tag returns, strandings, and nesting surveys, NMFS (2001a) 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 1988; NMFS and USFWS 1998d). Females usually breed every two to three years, but can vary from one to seven years (Dodd 1988; 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 to 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 to 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 1998h; 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 1988; Hildebrand 1983; Weishampel et al. 2006; Williams-Walls et al. 1983).

4.2.14.5 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 1988; NMFS 2001a; Rees and Margaritoulis 2004). Nest temperature seems to drive sex determination. Along the Florida coast, males primarily derive from earlier-season nesting (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 (1988) 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 to 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 temperatures for loggerheads on the northern Gulf of Mexico (Lamont and Fujisaki 2014).

4.2.14.6 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 1988; Wallace et al. 2009). Loggerheads in the deep, offshore waters of the western North Pacific feed on jellyfish, salps, and other gelatinous animals (Dodd 1988; 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).

4.2.14.7 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 (98 ft) (Sakamoto et al. 1993). Routine dives can last 4 to 172 minutes (Byles 1988; Renaud and Carpenter 1994; Sakamoto et al. 1990). The maximum-recorded dive depth for a post-nesting female was over 230 m (755 ft), although most dives are far shallower (9 to 21 m [29.5 to 69 ft]) (Sakamoto et al. 1990). Loggerheads tagged in the Pacific over the course of five months showed that about 70 percent of dives are very shallow (less than 5 m [16.4 ft]) 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° North latitude and another of 17° C at 32° North latitude. In the Mediterranean, dives of over 300 minutes have been recorded in association with depressed water temperatures and are proposed as an overwintering strategy (Luschi et al. 2013).

4.2.14.8 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. 1999b; Lenhardt 1994; Lenhardt 2002; Moein Bartol and Ketten 2006; Ridgway et al. 1969). Bartol et al. (1999b) 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).

4.2.14.9 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 individuals (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.

4.2.14.10 Natural Threats

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, and to a lesser extent by killer whales.

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).

4.2.14.11 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, anthropogenic threats appear to disproportionally impact larger (more fecund) loggerheads (Bellido et al. 2010).

The major factors inhibiting loggerhead 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). Each year, various fisheries capture about 2,000 loggerhead sea turtles in Pamlico Sound, of which almost 700 die. 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). 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 ten 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). 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 to 4,000 deaths per year, or about ten 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. 2008). 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, perfluorooctanesulfonic acid (PFOS), perfluorooctanoic acid (PFOA), DDT, and PCBs (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).

4.2.14.12 Critical Habitat

On July 10, 2014, NMFS designated critical habitat for loggerhead turtles in coastal and offshore waters of the U.S. Atlantic and Gulf of Mexico from North Carolina to Mississippi (79 FR

39856). Since loggerhead sea turtle critical habitat does not occur in the action area for SURTASS LFA sonar, it will not be addressed further in this opinion and conference report.

4.2.15 Olive Ridley Sea Turtle

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

4.2.15.1 Distribution

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

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

In the eastern Pacific, olive ridley turtles nest along the Mexico and Central American coast, with large nesting aggregations occurring at a few select beaches located in Mexico and Costa Rica. Few turtles nest as far north as southern Baja California, Mexico (Fritts et al. 1982) or as far south as Peru (Brown and Brown 1982). The post-nesting migration routes of olive ridleys traversed thousands of kilometers of deep oceanic waters, ranging from Mexico to Peru, and more than 3,000 km (1,620 nmi) out into the central Pacific (Plotkin 2007). Although they are the most abundant north Pacific sea turtle, surprisingly little is known of the oceanic distribution and critical foraging areas of Pacific ridley turtles.

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

4.2.15.2 Population Structure

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

4.2.15.3 Natural Threats

The various habitat types that olive ridley turtles occupy throughout their lives exposes these sea turtles to a wide variety of natural threats. The beaches on which olive ridley turtles nest and the nests themselves are threatened by hurricanes and tropical storms as well as the storm surges, sand accretion, and rainfall that are associated with hurricanes. Hatchlings are hunted by predators like herons, gulls, dogfish, and sharks. Adult olive ridley turtles are also killed by sharks and other large, marine predators. In addition, some female olive ridleys nesting in Costa Rica have been found afflicted with the fibropapilloma disease (Aguirre et al. 1999).

4.2.15.4 Anthropogenic Threats

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

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

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

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

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

4.2.15.5 Status and Trends

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

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

At a nesting site in Costa Rica, an estimated 0.2 percent of 11.5 million eggs laid during a single arribada produced hatchlings (NMFS and USFWS 1998g). At Playa La Flor, the second most important nesting beach for Pacific ridleys on Nicaragua, Ruiz (Ruiz 1994) documented six arribadas (defined as 50 or more females resting simultaneously). The main egg predators were domestic dogs and vultures (*Coragyps atratus* and *Cathartes aura*).

In the western Pacific, information on the size of olive ridley nesting aggregations are limited although they do not appear to be recovering (with the exception of the nesting aggregation at Orissa, India). There are a few sightings of olive ridleys from Japan but no reports of egg-laying. Similarly, there are no nesting records from China, Korea, the Philippines, Taiwan, Viet Nam, or Kampuchea, and nesting records in Indonesia are not sufficient to assess population trends (Eckert 1993b; Suwelo 1999). In Thailand, olive ridleys occur along the southwest coast, on the

Surin and Similan islands, and in the Andaman Sea. On Phra Thong Island, on the west coast of Thailand, the number of nesting turtles has declined markedly from 1979 to 1990.

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

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

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

4.2.15.6 Diving and Social Behavior

Although olive ridley turtles are probably surface feeders, they have been caught in trawls at depths of 80 to 110 m (262 to 361 ft) (NMFS and USFWS 1998g), and a post-nesting female reportedly dove to a maximum depth of 290 m (951 ft). The average dive length for an adult female and adult male is reported to be 54.3 and 28.5 minutes, respectively (Plotkin 1994, in Lutcavage and Lutz 1997).

4.2.15.7 Vocalizations and Hearing

Sea turtles do not appear to use sound for communication, and there are no published recordings of olive ridley turtle vocalizations. There is no information on olive ridley turtle hearing. However, we assume that their hearing sensitivities will be similar to those of green, hawksbill, leatherback, and loggerhead turtles, whose best hearing sensitivity is in the low frequency range, with maximum sensitivity below 400 Hz and an upper hearing range not likely to exceed 2,000 Hz.

4.2.15.8 Critical Habitat

Olive ridley sea turtle critical habitat has not been designated.

4.2.16 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 900 kg (2,000 lbs). The leatherback is the only sea turtle that lacks a hard, bony shell. A leatherback's carapace is approximately 4 cm (1.5 in) 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 eight 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 in) in length, with fore flippers as long as their bodies, and weigh approximately 40 to 50 g (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.

4.2.16.1 Populations

Leatherbacks are divided 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.

Leatherback 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 the Pacific Ocean, leatherbacks are found from tropical waters to polar waters of Alaska and are 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 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). In Fiji, Thailand, and Australia, leatherback sea turtles have only been known to nest in low densities and at scattered sites.

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 to 65 days (Lux et al. 2003). Limited nesting also occurs along Nicaragua, Panama, El Salvador, Vanuatu, and Guatemala.

Genetic studies have identified three distinct Pacific Ocean 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 2006; Dutton et al. 2000b; Dutton et al. 2006; Dutton 2007). The genetically distinct Malaysia nesting population likely is extirpated (Chan and Liew 1996; Dutton et al. 1999; Dutton 2006).

4.2.16.2 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, 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 km (5,400 nmi) (Eckert 1998). In the North Atlantic Ocean, leatherback turtles regularly occur in deep waters (greater than 100 m [328 ft]), and an aerial survey study in the north Atlantic sighted leatherback turtles in water depths ranging from 0.9 to 4,151 m (3 to 13,618 ft), with a median sighting depth of 40.1 m (131.6 ft), and in waters ranging from 7 to 27.2° C (CETAP 1982). 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° North and 47° South 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 2007c). 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. 1998). Leatherbacks from nesting beaches in the Indo-Pacific region have been tracked migrating thousands of kilometers through the North Pacific Transition Zone to summer foraging grounds off the coast of northern California (Benson et al. 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 2007c). 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.

4.2.16.3 Migration and Movement

Leatherback turtles are capable of migrating long distances throughout open ocean convergence zones, upwelling areas, along continental margins, and in archipelagic waters (Eckert 1998; Eckert 1999a; Morreale et al. 1994a). In a single year, a leatherback may swim more than 9,600 km (5,184 nmi) between nesting and foraging areas throughout ocean basins (Benson et al. 2007a; Dutton et al. 2007; Eckert 1998; 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, which assist in transporting individuals (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 more southern waters during winter and spring. Another strategy used by six individuals was similar to this, but instead of a southward movement in late fall, individuals overwintered in northern latitudes (30 to 40º North, 25 to 30º West) 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 (Fossette et al. 2009b). 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 to 50º North (Fossette et al. 2009b). Individuals nesting in Caribbean Islands migrate to foraging areas off Canada (Richardson et al. 2012).

Females tracked from nesting beaches in Brazil remained 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 in southeastern U.S. waters year-round, with peak abundance in summer (TEWG 2007). In spring, leatherback turtles appear to be concentrated in coastal waters, while other times of the year they are spread out to at least the Gulf Stream. From August 2009 through August 2010 off Jacksonville, Florida, surveys sighted 48 leatherback turtles, while simultaneous vessel surveys sighted four leatherback 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. 2000b; 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 (Collard 1990).

Satellite tracking data reveal that female leatherbacks leaving Mexican and Central American nesting beaches migrate towards the equator and into Southern Hemisphere waters, some passing the Galápagos Islands and dispersing south of 10º South (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 five to seven months to arrive on foraging grounds from nesting locations (Benson et al. 2011). Additional foraging occurs in temperate locations in waters along the U.S. Pacific coast; individuals take 10 to 12 months to migrate to northeastern Pacific waters (Benson et al. 2011). 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. 2011).

4.2.16.4 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, eddies, current boundaries, and coastal retention areas (Benson et al. 2011; 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, presumably associated with foraging opportunities (Dodge et al. 2014). Aerial surveys off the western U.S. have shown that leatherbacks occur in greater densities in continental slope waters than in 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 North in the Atlantic appear to be popular foraging locations (Fossette et al. 2009b). 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. 2011). 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 2,000 m (6,562 ft) 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, the 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 to 3,000 m (656 to 9,843 ft) isobaths from Point Arena to Point Sur, California and waters between the coastline and the 3,000 m 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).

4.2.16.5 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 to 6 years (Rhodin 1985) to 13 to 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 et al. 2009). Female leatherbacks nest frequently (up to 13, average of five to seven nests per year and about every two to three 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 (greater than 100 eggs (Eckert et al. 2012). However, up to approximately 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 to 75 days before hatching.

4.2.16.6 Sex Ratio

Leatherback sex determination is affected by nest temperature, with higher temperatures producing a greater proportion of females (Mrosovsky 1994; Witzell et al. 2005). 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.

4.2.16.7 Feeding

Leatherbacks may forage in high-invertebrate prey density areas formed by favorable oceanographic 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 (gelatinous zooplankton), 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 2006; Dutton et al. 2000a; Dutton et al. 1998). 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 and Oregon (Eisenberg and Frazier 1983; Stinson 1984c).

4.2.16.8 Diving

Leatherbacks are champion deep divers among sea turtles with a maximum recorded dive of over 4,000 m (13,123 ft) (Eckert et al. 1989; López-Mendilaharsu et al. 2009). Dives are typically 50 to 84 m (164 to 276 ft) and 75 to 90 percent of time duration is above 80 m (262 ft) (Standora et al. 1984). Leatherbacks off South Africa were found to spend less than one percent of their dive time at depths greater than 200 m (656 ft) (Hays et al. 2009). Dive durations are impressive, topping 86 minutes, but routinely only last one to 14 minutes (Eckert et al. 1989; Eckert et al. 1996; Harvey et al. 2006; López-Mendilaharsu et al. 2009). Most dive 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 to 68 percent of their time underwater, diving to a mean depth of 19 m (62 ft) for 7.4 minutes (Southwood et al. 1999). Off St. Croix, adult females dove to a mean depth of 61.6 m (202.1 ft) for an average of 9.9 minutes, and spent an average of 4.9 minutes at the surface (Eckert et al. 1989). During shallow dives in the South China Sea, dives averaged 6.9 to 14.5 minutes, with a maximum of 42 minutes (Eckert et al. 1996). Off central California, leatherbacks dove to 20 to 30 m (66 to 98 ft) with a maximum of 92 m (302 ft) (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). In a study comparing diving patterns during foraging versus travelling, leatherbacks dove shallower (mean of 53.6 m [175.9 ft]) and moved more slowly (17.2 km per day) while in foraging areas while travelling to or from these areas (81.8 m [268 ft] and 51.0 km [27.5 nmi] per day) (Fossette et al. 2009b).

4.2.16.9 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. 1999b; Lenhardt 1994; Lenhardt 2002; Moein Bartol and Ketten 2006; Ridgway et al. 1969). To date, only one study has investigated hearing of the leatherback turtle (Piniak 2012). Piniak (2012) measured hearing of hatchlings in water and in air, and observed reactions to low frequency sounds, with responses to stimuli occurring between 50 Hz and 1.6 kHz in air and between 50 Hz and 1.2 kHz in water (lowest sensitivity recorded was 93 dB re: 1 µPa at 300 Hz).

4.2.16.10 Status and Trends

Leatherback 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 abundance of nests has 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 2001a; NMFS 2001b).

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 through 2009). Annual survival probability (ca. 0.85) was constant over the ten-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 nine 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 2,700 to 4,500 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 one percent of that in 1950s (Chan and Liew 1996). 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 less than 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 2003). 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 NOAA satellite tracking of leatherback turtles indicates sea turtles departing from regional nesting habitats and transiting through the waters of the SURTASS LFA sonar action area (Benson et al. 2007a; Benson et al. 2011). As for nearshore waters, Eldredge (2003) noted a rescue in 1978 of a 112.9 kg (249 lb) 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 2010). 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.

4.2.16.11 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. 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). Lost timber on beaches and beachcast logs can trap adult females nesting on Gabon beaches, resulting in mortality (Ikaran 2013).

4.2.16.12 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 1991). 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, since plastic resembles jellyfish and other gelatinous zooplankton, leatherback's favored prey, 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 one or two 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 five 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 eight 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 three-year period starting in 2007 (NMFS 2004a). 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 100 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 NMFS Northeast Fisheries

Science Center (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 2001a).

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. 2010b; Santidrián Tomillo et al. 2007; Tomillo et al. 2008). For example, during the 2012 nesting season, 55 percent (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).

4.2.16.13 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 (600 ft) isobath to mean high tide level between 17° 42’12” North and 65° 50’00” West (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 Pacific Ocean waters along Washington and Oregon (Cape Flattery to Cape Blanco; 64,760 km^2 [18855 nmi^2]) and California (Point Arena to Point Arguello; 43,798 km^2 [12,769 nmi^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.

Since leatherback sea turtle critical habitat does not occur in the action area for SURTASS LFA sonar, it will not be addressed further in this opinion and conference report.

4.2.17 U.S. Navy-estimated Species Abundances for ESA-listed Marine Mammal Stocks

The U.S. Navy estimated marine mammal stock abundances for ESA-listed species as part of their modeling effort (see Section 3.8.1). Where U.S. Navy's mission areas coincide with areas in which NMFS has made stock or DPS delineations, the U.S. Navy has used those (see in particular those for the Hawaii and offshore Guam mission areas). For mission areas in much of the western North Pacific Ocean part of the action area for SURTASS LFA sonar, the U.S. Navy delineated geographically separate stocks according to the best literature and information available. The marine mammal stock abundances and densities used by the U.S. Navy represent the most up-to-date estimates for the stocks as defined herein and may not be identical to those in the literature as discussed above for ESA-listed entities or by ocean basin.

Table 18. Estimated abundance of each ESA-listed marine mammal stock that may be affected by the operation of SURTASS LFA sonar.

Species	Marine Mammal Stock or DPS	Estimated Number in Stock
East of Japan		
Blue whale	WNP	9,250
Fin whale	WNP	9,250
Humpback whale	WNP	1,328
North Pacific right whale	WNP	922
Sei whale	NP	77,000
Sperm whale	NP	102,112
North Philippine Sea		
Blue whale	WNP	9,250
Fin whale	WNP	9,250
Humpback whale	WNP	1,328
North Pacific right whale	WNP	922
Sperm whale	NP	102,112
West Philippine Sea		
Blue whale	WNP	9,250
Fin whale	WNP	9,250
Humpback whale	WNP	1,328
Sperm whale	NP	102,112
Offshore Guam		
Blue whale	WNP	9,250
Fin whale	WNP	9,250
Humpback whale	WNP	1,328
Sei whale	NP	7,000
Sperm whale	NP	102,112

Species	Marine Mammal Stock or DPS	Estimated Number in Stock
Sea of Japan		
Fin whale	WNP	9,250
North Pacific right whale	WNP	922
Western North Pacific gray whale	WNP	140
Sperm whale	NP	102,112
East China Sea		
Fin whale	ECS	500
North Pacific right whale	WNP	922
Western North Pacific gray whale	WNP	140
Sperm whale	NP	102,112
Spotted seal	Southern	1,000
South China Sea		
Fin whale	WNP	9,250
Humpback whale	WNP	1,328
North Pacific right whale	WNP	922
Western North Pacific gray whale	WNP	140
Sperm whale	NP	102,112
Offshore North Pacific (25 to 40° North) (Offshore Japan)		
Blue whale	WNP	9,250
Fin whale	WNP	9,250
Humpback whale	WNP	1,328
Sei whale	NP	8,600
Sperm whale	NP	102,112
Hawaiian monk seal	Hawaiian	1,153153
Offshore North Pacific (10 to 25° North) (Offshore Japan)		
Blue whale	CNP	9,250
Fin whale	WNP	9,250
Humpback whale	WNP	1,328
Sei whale	NP	77,000
Sperm whale	NP	102,112
Hawaii North		
Blue whale	CNP – Hawaii	81
Fin whale	Hawaii	58
Sei whale	Hawaii	178
Humpback whale	CNP - Hawaii	10,103
Sperm whale	Hawaii	3,354
MHI Insular False Killer Whale	MHI Insular	151

Species	Marine Mammal Stock or DPS	Estimated Number in Stock
Hawaiian monk seal	Hawaii	1,153
Hawaii South		
Blue whale	CNP – Hawaii	81
Fin whale	Hawaii	58
Sei whale	Hawaii	178
Humpback whale	CNP - Hawaii	10,103
Sperm whale	Hawaii	3,354
MHI Insular False Killer Whale	MHI Insular	151
Hawaiian monk seal	Hawaiian	1,153
Arabian Sea		
Blue whale	North Indian	3,432
Fin whale	Indian	1,716
Humpback whale	Arabian Sea	200
Sperm whale	North Indian	24,446
Andaman Sea		
Blue whale	North Indian	3,432
Fin whale	Indian	1,716
Sperm whale	North Indian	24,446
Northwest of Australia		
Blue whale	South Indian	1,657
Fin whale	South Indian	38,185
Humpback whale	West Australia	14,000
Sperm whale	South Indian	24,446
Northeast of Japan		
Blue whale	WNP	9,250
Fin whale	WNP	9,250
Humpback whale	WNP	1,328
North Pacific right whale	WNP	922
Sei whale	NP	7,000
Western North Pacific gray whale	WNP	140
Sperm whale	NP	102,112
Steller sea lion	Western	82,516

5 ENVIRONMENTAL BASELINE

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

5.1 Climate Change

The 2014 *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 (246 ft) have warmed by 0.11° C per decade over the period 1971 to 2010) (IPCC 2014). Global mean sea level rose by 0.19 m (0.62 ft) 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 and conference report, 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. 1999a; Payne et al. 1986; Payne et al. 1990). Pecl and Jackson (2008) predicted climate change will likely result in squid that hatch out smaller and earlier, undergo faster growth over shorter life-spans, and mature younger at a smaller size. This could have significant negative consequences for species such as sperm whales, whose diets can be dominated by cephalopods. For ESA-listed species that undergo long migrations, if either prey availability or habitat suitability is disrupted by changing ocean temperature regimes, the timing of migration can change or negatively impact population sustainability (Simmonds and Eliott. 2009).

Changes in global climatic patterns are expected to have profound effects on coastlines worldwide, potentially having significant consequences for the species considered in this opinion and conference report that are partially dependent on terrestrial habitat areas (i.e., sea turtles). For example, rising sea levels are projected to inundate some sea turtle nesting beaches (Caut et al. 2009; Wilkinson and Souter 2008), change patterns of coastal erosion and sand accretion that are necessary to maintain those beaches, and increase the number of turtle nests destroyed by tropical storms and hurricanes (Wilkinson and Souter 2008). The loss of nesting beaches may have catastrophic effects on global sea turtle populations if they are unable to colonize new beaches, or if new beaches do not provide the habitat attributes (e.g., sand depth, temperature regimes, refuge) necessary for egg survival. Additionally, increasing temperatures in sea turtle nests, as is expected with climate change, alters sex ratios, reduces incubation times (producing smaller hatchlings), and reduces nesting success due to exceeded thermal tolerances (Fuentes et al. 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. 2004; 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).

This is not an exhaustive review of all available literature regarding the potential impacts of climate change to the species considered in this opinion and conference report. 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 and conference report, a range of consequences are expected, ranging from beneficial to catastrophic. Given a lack of available information within the context of the temporal scale of the action, specific climate change related impacts on the species evaluated in this opinion and conference report are speculative, cannot be meaningfully assessed, and will not be considered further.

5.2 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 ESA-listed species in the action areaarea. 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 ESA-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 thatnine 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. 2010b). 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. 2010b) 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 10,886,216 to 21,772,433 kg (12,000 to 24,000 U.S. tons) of plastic debris a year (Davison and Asch 2011).

5.3 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 non-targeted 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). 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).

5.4 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, CNMI, and Hawaii, 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.

5.5 Whaling

Large whale population numbers in the action area have historically been impacted by commercial exploitation, mainly in the form of whaling. 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 19. lists the reported catches of North Pacific endangered whale species considered in this opinion and conference report and the year in which the IWC issued a moratorium on harvest of that species.

Table 19. Reported catch of ESA-listed whales considered in this opinion and conference report, in the North Pacific Ocean.

Species	Estimated Total Catch	Data Years	Source	IWC Moratorium
Blue whale	9,500 whales	1910 to 1965	(Ohsumi and Wada 1972)	1966
Fin whale	46,000 whales	1919 to 1945	(C. Allison, IWC, pers. comm.; cited in : (Carretta et al. 2014)	1976
Humpback whale	15,000 whales	1919 to 1987	(Tonnessen and Johnsen 1982); C. Allison, IWC unpubl. Data; cited in: (Carretta et al. 2014)	1966
Sei whale	61,500 whales	1947 to 1987	(C Allison, IWC, pers. comm.(Allison 2007)	1976
Sperm whale	258,000 whales	1947 to 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 scientific permit 1985-2013 (Table 20.). Some of the whales killed in these fisheries are likely part of the same populations of whales occurring within the action area for this consultation.

Table 20. Catches taken in the Pacific Ocean by commercial, aboriginal, and scientific permit whaling since 1985. Note that the large majority of these catches were taken in the Northwest Pacific Ocean by either Japan or Russia (USSR prior to 1992). Data compiled from the IWC website (iwc.int/home; originally accessed on January 24, 2015; updated July 29, 2016).

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**	**3,542**	**1,179**

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 and conference report, 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 humpback 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 Vessel Strike

Vessel strike is a significant concern for the recovery of ESA-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, unlike other maritime entities, the U.S. 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. 2010). 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 per hour; most vessels move far faster than this in open water (Hazel and Gyuris 2006; Hazel et al. 2007; Work et al. 2010). 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.

5.7 Scientific Research

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 July 29, 2016, there were 38 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 July 29, 2016, there were currently 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. Pinniped research includes capture, biopsy, nail clip, blood sampling, and anesthesia, among other techniques. As of July 29, 2016, there were currently six permits in the Pacific Ocean authorizing research on Hawaiian monk seals. Most research involves sub-lethal take (e.g., capture/release), but ten unintentional mortalities are permitted annually, as well as 20 instances of intentional mortality. Lethal take of male Hawaiian monk seals has been authorized in specific instances of mobbing. The removal of specific males involved in mobbing preserves the health of female and young individuals that will provide greater contributions to the survival and recovery of the species. As of July 29, 2016, there were six permits in the Pacific Ocean authorizing research on Western DPS Steller sea lions. Most authorized take is sub-lethal (e.g., capture/release), but 14 untintentional mortalities are authorized annually.

5.8 Vessel Approaches – Commercial and Private Marine Mammal Watching

As of 2010, commercial whale watching was a $1 billion global industry per year (Lambert et al. 2010). Private vessels may partake in this activity as well. Whale watching, particularly of humpback whales, is extensive in Hawaiian waters during the winter. NMFS has issued certain regulations and guidelines relevant to whale watching. For example, under 50 CFR 224.103, except as authorized under the MMPA and ESA, federal regulations prohibit approaching (by any means) humpback whales within 100 yards when on or in the water, and within 305 m (1,000 ft) when operating an aircraft within 370 km (200 nmi) of the Islands of Hawaii. As noted previously, many of the cetaceans considered in this opinion and conference report are highly migratory, so may also be exposed to whale watching activity occurring outside of the action area.

Although considered by many to be a non-consumptive use of marine mammals with economic, recreational, educational and scientific benefits, marine mammal watching is not without potential negative impacts. Whale watching has the potential to harass whales by altering feeding, breeding, and social behavior or even injure them if the vessel gets too close or strikes the whale. Preferred habitats may be abandoned if disturbance levels are too high. Animals may also become more vulnerable to vessel strikes if they habituate to vessel traffic (Swingle et al. 1993; Wiley et al. 1995). Several investigators have studied the short term effects of whale watch vessels on marine mammals (Amaral and Carlson 2005; Au and Green 2000a; Corkeron 1995; Erbe 2002b; Felix 2001; Magalhaes et al. 2002; Richter et al. 2003a; Scheidat et al. 2004; Simmonds 2005; Watkins 1986; Williams et al. 2002b). The whale's behavioral responses to whale watching vessels depended on the distance of the vessel from the whale, vessel speed, vessel direction, vessel noise, and the number of vessels. In some circumstances, the whales did not respond to the vessels, but in other circumstances, whales changed their vocalizations, surface time, swimming speed, swimming angle or direction, respiration rates, dive times, feeding behavior, and social interactions. Although numerous short-term behavioral responses to whale watching vessels are documented, little information is available on whether long-term negative effects result from whale watching (NMFS 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. Christiansen et al. (2014) suggested that the whale watching industry, in its current state, is likely not having any long-term negative effects on vital rates.

It is difficult to precisely quantify or estimate the magnitude of the risks posed to marine mammals in general from vessel approaches. Given that SURTASS LFA sonar sound fields greater than 180 dB re: 1 µPa (rms) will not occur in areas with 22 km (12 nmi) of land, few whale watching boats would be expected to co-occur with the SURTASS LFA sonar vessels.

5.9 Vessel Noise and Commercial Shipping

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

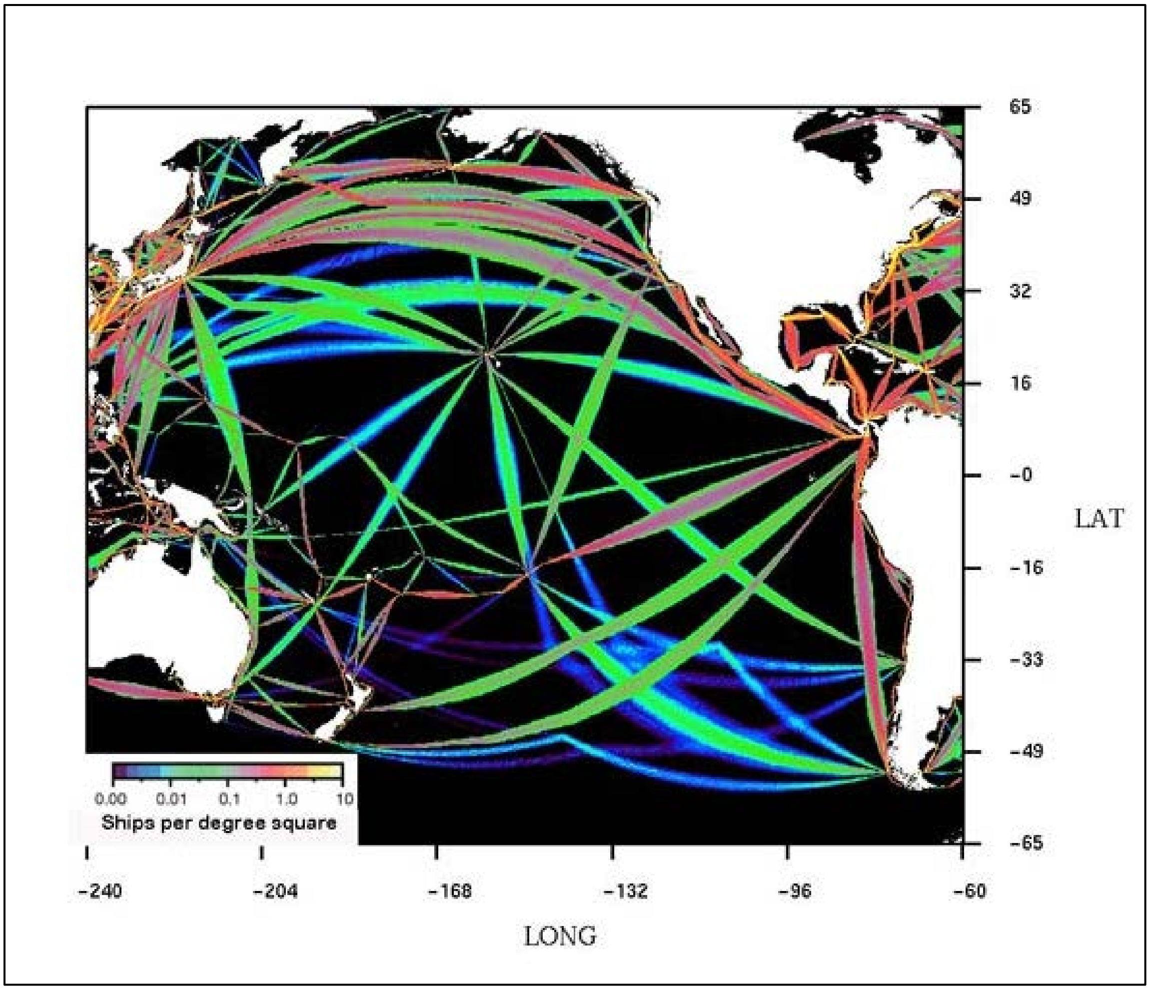

Figure 8. Estimated number of ships per one degree latitude by one degree longitude for the Pacific Ocean. Courtesy of R. Heitmeyer (U.S. Naval Research Laboratory). (Adapted from (Hildebrand 2009a).

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

Measurements made over the period 1950 through 1970 indicated LF (50 Hz) ship traffic noise in the eastern North Pacific and western North Atlantic Oceans was increasing by 0.55 dB per year. Data obtained in the northeast Pacific from 1978 to 1986 suggest the 0.55 dB per year increase seen in the early data continued to around 1980, but then slowed to about 0.2 dB per year (Chapman and Price 2011). Data obtained in the northeast Pacific from 1978 to 1986 suggest the 0.55 dB per year increase seen in the early data continued to around 1980, but then slowed to about 0.2 dB per year (Chapman and Price 2011). Hildebrand (2009) documented a three dB per decade increase in ambient noise in the 40 Hz band starting in the late 1950s (Hildebrand 2009a). Similar trends were documented in the Indian Ocean with the most prominent increases in ocean noise occurring in the 85 to 105 Hz band. The authors noted that this increase was consistent with concurrent increases in shipping, wind speed, wave height, and blue whale abundance in the Indian Ocean (Miksis-Olds et al. 2013).

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

Many researchers have described behavioral responses of marine mammals to the sounds produced by boats and ships, as well as other noise sources such as helicopters and fixed-wing aircraft, and dredging and construction. Most observations have been limited to short term behavioral responses, which included temporary cessation of feeding, resting, or social interactions; however, habitat abandonment can lead to more long-term effects which may have implications at the population level. Masking may also occur, in which an animal may not be able to detect, interpret, and/or respond to biologically relevant sounds. Masking can reduce the range of communication, particularly long-range communication, such as that for blue and fin whales. This could have a variety of implications for an animal’s fitness including, but not limited to, predator avoidance and the ability to reproduce successfully (MMC 2007). Recent scientific evidence suggests that marine mammals, including blue and fin whales, compensate for masking by changing the frequency, source level, redundancy, or timing of their signals, but the long-term implications of these adjustments are currently unknown (McDonald et al. 2006a; Parks 2003; Parks 2009a).

5.10 Ongoing U.S. Navy Pacific Fleet Training and Testing Activities – Hawaii Range Complex

From January 8, 2009 to December 25, 2015, the U.S. Navy conducted 16 individual major training events (MTEs) in the Hawaii Range Complex (

Table 21.). SURTASS LFA sonar operations were not involved in any of the Hawaii Range Complex MTEs.

Table 21. Hawaii Range Complex MTEsummary.

Exercise Type	Jan 8, 2009 – Jan 14, 2010	Jan 15, 2010 – Jan 14, 2011	Jan 15, 2011 – Jan 14, 2012	Jan 15, 2012 – Jan 14, 2013	Jan 15, 2013 – Dec 25, 2013	Dec 26, 2013 – Dec 25, 2014	Dec 26, 2014 – Dec 25, 2015	7-Year Total
USWEX	2	0	6	1	1	2	0	12
RIMPAC	0	1	0	1	0	1	0	3
IAC	0	0	0	0	0	0	1	1
Total	**2**	**1**	**6**	**2**	**1**	**3**	**1**	**16**

USWEX = Undersea Warfare Exercise
RIMPAC = Rim of the Pacific
IAC = Integrated Antisubmarine Warefare Course

5.10.1 Rim of the Pacific Exercise

The most recent biennial RIMPAC Exercise was completed August 1, 2014 and involved the navies of 22 nations, more than 49 ships, six submarines, more than 200 aircraft and 25,000 personnel that operated in and around the Hawaiian Islands and Southern California. U.S. Navy units from Australia, Brunei, Canada, Chile, Colombia, France, India, Indonesia, Japan, Malaysia, Mexico, Netherlands, New Zealand, Norway, the People’s Republic of China, Peru, the Republic of Korea, the Republic of the Philippines, Singapore, Tonga, the United Kingdom and the United States participated. SURTASS LFA sonar operations were not involved in any of the Hawaii Range Complex RIMPAC Exercises.

Figure 9. Pacific Ocean (July 25, 2014) Forty-two ships and submarines representing 15 international partner nations maneuver into close formation during RIMPAC 2014. (U.S. Navy photo by Mass Communication Specialist 1st Class Shannon Renfroe/Released)

5.10.2 Marine Animal Sightings

There were at least 1,784 marine mammals and sea turtles during MTEs in the Hawaii Range Complex from January 8, 2009 to December 25, 2015 (Table 22.).

Table 22. Hawaii Range Complex sighted marine mammals and sea turtlesturtles.

Marine Animal Types	Jan 8, 2009 – Jan 14, 2010	Jan 15, 2010 – Jan 14, 2011	Jan 15, 2011 – Jan 14, 2012	Jan 15, 2012 – Jan 14, 2013	Jan 15, 2013 – Dec 25, 2013	Dec 26, 2013 – Dec 25, 2014	Dec 26, 2014 – Dec 25, 2015	6-Year Total
	Estimated Number of Animals Sighted While Sonar Active							
Dolphin	0	3	45	208	0	50	0	306
Whale	3	0	12	13	0	1	1	30
Pinniped	0	0	0	0	0	0	0	0
Sea Turtle	0	0	0	0	0	0	0	0
Unknown	1	0	1	3	5	1	0	11
Subtotal while Active	4	3	58	224	5	52	1	347
	Estimated Number of Animals Sighted While Sonar Passive							
Dolphin	0	255	33	228	0	262	0	778
Whale	133	34	58	155	3	140	4	527
Pinniped	1	0	0	1	0	0	0	2
Turtle	0	25	0	1	0	0	0	26
Unknown	7	0	4	1	1	91	1	105
Subtotal while Passive	141	314	95	386	4	493	5	1,438
Total	**145**	**317**	**153**	**610**	**9**	**545**	**5**	**1,784**

5.10.3 Hawaii Range Complex Mitigation Events

There were 27 total mitigation events where sonar was powered down, shut down, or delayed due to the sighting of marine mammals or sea turtles during MTEs from January 8, 2009 to December 25, 2015.

5.11 Ongoing U.S. Military Training and Testing Activities in the Mariana Islands Range Complex

From August 2010 through August 2015, the U.S. Navy (Pacific Fleet) conducted three MTEs in its Mariana Islands Range Complex (MIRC) and performed marine species monitoring under the MMPA LOA for at-sea training in the MIRC (Table 23.). MTE and monitoring data were provided by the U.S. Navy in the Comprehensive Exercise and Marine Species Monitoring Report for the U.S. Navy's MIRC. The majority of the training and testing activities the U.S. Navy conducts in the MIRC 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. One SURTASS LFA sonar vessel participated in the 2012 joint MTE. A SURTASS vessel also participated in the 2014 joint MTE.

Table 23. Summary of MTEs in MIRC.

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

There were an estimated 62 marine mammals during MTEs in the MIRC from August 12, 2010 to August 3, 2015 (Table 24.).

Table 24. Summary of Sightings during Major Training Exercises in MIRC.

Marine Species	Aug 12, 2010 – Feb 15,	Feb 16, 2011 – Feb 15,	Feb 16, 2012 – Feb 15, 2013	Feb 16, 2013 – Feb 15, 2014	Feb 16, 2014 – Aug 3, 2015	Reporting Period Total
Estimated Number of Animals Sighted While Sonar Active						
Dolphin	0	0	0	0	0	0
Whale	4	0	0	0	4	8
Pinniped	0	0	0	0	0	0
Turtle	0	0	0	0	0	0
Generic	1	0	0	0	5	6
Subtotal while Active	**5**	**0**	**0**	**0**	**9**	**14**
Estimated Number of Animals Sighted While Sonar Passive						
Dolphin	25	0	8	0	1	34
Whale	9	0	0	0	5	14
Pinniped	0	0	0	0	0	0
Turtle	0	0	0	0	0	0
Generic	0	0	0	0	0	0
Subtotal while Passive	34	0	8	0	6	48
Total	**39**	**0**	**8**	**0**	**15**	**62**

There were seven 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 August 12, 2010 to August 3, 2015 (Table 25). All of these sightings were observed within 4,000 yards of the U.S. Navy unit. The U.S. Navy's unclassified annual exercise reports from 2010 through 2015 contain tables listing all marine mammals sighted during that reporting year and the range of the sighting.

Table 25.Summary of mitigation events during MTEs in MIRC.

<table>
<tr><th>Marine Animal Species</th><th>Range of Detection (Yards, <200, 200-500, 500-1,000, 1,000-2,000, >2,000)</th><th>Mitigation Measure Implemented</th><th colspan="2">Un-required Mitigation (Yes/No)</th></tr>
<tr><td colspan="5">12 August 12, 2010 – February 15, 2011</td></tr>
<tr><td>Generic</td><td>Acoustic detection</td><td>Sonar shut down</td><td colspan="2">Yes</td></tr>
<tr><td>Whale</td><td><200</td><td>Sonar shut down</td><td colspan="2">No</td></tr>
<tr><td>Whale</td><td>200-500</td><td>Sonar shut down</td><td colspan="2">No</td></tr>
<tr><td>Whale</td><td>>2,000</td><td>Sonar shut down/maneuvered</td><td colspan="2">Yes</td></tr>
<tr><td colspan="5">February 16, 2011 – February 15, 2012</td></tr>
<tr><td colspan="5">No mitigation events during this period</td></tr>
<tr><td colspan="5">February 16, 2012 – February 15, 2013</td></tr>
<tr><td colspan="5">No mitigation events during this period</td></tr>
<tr><td colspan="5">February 16, 2013 – February 15, 2014</td></tr>
<tr><td colspan="5">No mitigation events during this period</td></tr>
<tr><td colspan="5">February 16, 2014 – August 3, 2015</td></tr>
<tr><td>Generic</td><td><200</td><td>Sonar shut down</td><td colspan="2">No</td></tr>
<tr><td colspan="2">Generic</td><td>500-1,000</td><td>Sonar shut down</td><td>No</td></tr>
<tr><td colspan="2">Generic</td><td>1,000-2,000</td><td>Sonar shut down</td><td>No</td></tr>
</table>

5.12 Ongoing SURTASS LFA Sonar Routine Training, Testing, and Military Operations

SURTASS LFA sonar is a coherent low frequency signal with a duty cycle of less than 20 percent, operating for a maximum of only 432 hours per year for each system or a total of 32 days per year. This compares to an approximate 21.9 million days per year for the world's shipping industry (presuming an 80 percent activity rate all the time). Thus, SURTASS LFA sonar noise would make up a very small part of the human-caused noise pollution in the ocean.

5.12.1 Previous and Ongoing MMPA Authorizations for SURTASS LFA Sonar Activities

The information below is a summary of quarterly and annual mission reports provided by the U.S. Navy as a requirement of the ESA and MMPA authorizations. Past operation of SURTASS LFA sonar in the western and central North Pacific Ocean over the thirteen plus year period spanning from 2002 through the end of the August 2015 LOA/ITS reporting period involved 173 completed missions conducted in slightly over 575days during which LFA sonar was transmitted for a total of 1,128 hours (Table 26). During those missions, 16 marine mammals or sea turtles were visually observed, eight marine animals were detected passive-acoustically, and 166 marine mammals/animals were detected active-acoustically by the HF/M3 sonar system. These combined detections of marine animals led to 176 suspensions/delays of LFA sonar transmissions, per the mitigation protocol for the operation of SURTASS LFA sonar (See Table 26). LFA sonar transmissions were also suspended or delayed during this period due to equipment faults, sonar wavetrain reconfiguration, or equipment changes.

During the most recent full reporting period which began on August 15, 2014 and was completed on August 14, 20152016, ten missions were conducted in the western North Pacific Ocean mission areas using four SURTASS LFA sonar systems.. In total during the 2014 to 2015 reporting period, the U.S. Navy conducted ten SURTASS LFA sonar missions over 22.9 days that resulted in total LFA sonar transmissions of 51.6 hours. Of the permitted 1,728 hours of LFA sonar transmit time for four SURTASS LFA sonar vessels, the U.S. Navy transmitted 2.9 percent of its permitted allocation. During the 51.6 hours of LFA sonar transmissions, and in accordance with the mitigation monitoring protocol for SURTASS LFA sonar, LFA sonar was suspended or delayed 11 times during the 2014 through 2015 annual LOA reporting period due to seven active acoustic and six visual detections of marine animals. No passive acoustic detections were reported in the mitigation/buffer zone. The visual detections included one sighting of a sea turtle and an unidentified whale, two pods of dolphins, and one sighting of a second unidentified whale. The two passive acoustic detections of marine mammal vocalizations that were evaluated to be outside of the mitigation/buffer zone were identified as Bryde’s and fin whales. No dead or injured marine animals were observed in conjunction with LFA sonar operations during the annual reporting period.

For the first two quarters of the current 2015 to 2016 reporting period, which commenced on August 15, 2015, three missions were conducted as of February 14, 2016 totaling 6.3 days and 14.6 hours of LFA sonar transmissions (Table 26). During the three missions, one visual detection of marine mammals was made, and two passive acoustic detections of marine animals were reported. The one visual detection was identified as whales and one of the two reported passive acoustic detections was evaluated to be a Bryde’s whale (*Balaenoptera edeni*) vocalization. Additionally, four HF/M3 sonar detections were made during the three missions. Per mitigation protocol, LFA sonar transmissions were suspended or delayed four times due to the visual and HF/M3 detections; LFA sonar transmissions were also suspended/delayed 62 times due to sonar technical issues and not due to detections of marine animals. No dead or injured marine animals were observed during any of the SURTASS LFA operations to date in the 2015 to 2016 effective period.

Both the historical and the most recent results (Table 26) 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.

Table 26. Summary of SURTASS LFA sonar missions and mitigation measure detections from 2002 through February 14, 2016.

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

	USNS *IMPECCABLE*	7			0	0	2	0	2
2011-2012	USNS *ABLE*	6	28.5	66.3	0[7]	1	1	0	3[8]
	USNS *EFFECTIVE*	5			0	0	0	0	0
	USNS *IMPECCABLE*	3			0	0	0	0	0
	USNS *VICTORIOUS*	1			0	0	0	0	0
2012-2013	USNS *ABLE*	3	24.4	47.3	0	0	0	0	0
	USNS *EFFECTIVE*	4			0	3	9	12	16
	USNS *IMPECCABLE*	2			0	0	0	0	12
	USNS *VICTORIOUS*	3			0	0	1	1	1
2013-2014	USNS *ABLE*	3	20.44	38.63	0	0	9	0	4
	USNS *EFFECTIVE*	3			0	0	9	0	0
	USNS *IMPECCABLE*	0			0	0	0	0	0
	USNS *VICTORIOUS*	1			0	0	1	0	1
2014-2015	USNS *ABLE*	1	22.9	51.6	0	0	1	2	0
	USNS *EFFECTIVE*	6			5	11	5	30	9
	USNS *IMPECCABLE*	2			1	18	1	23	1
	USNS *VICTORIOUS*	1			0	0	0	5	0
August 15, 2015 – February 14, 2016	USNS *ABLE*	2	6.3	14.6	0	0	0	50	0
	USNS *EFFECTIVE*	1			1	0	4	12	4
	USNS *IMPECCABLE*	0			0	2	0	0	0
	USNS *VICTORIOUS*	0			0	0	0	0	0
	Totals	**175**	**575.44**	**1128.05**	**19**	**39**	**162**	**239**	**299**

[1] *August to August*

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

[3] *Contact made when LFA sonar not transmitting*

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

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

[6] *Only one contact confirmed visually*

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

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

[9] *LFA suspended 5 times due to equipment faults*

[10] *Starting 2014, LFA sonar suspensions/delay for non-mitigation factors recorded. Includes HF/M3 non-availability as well as operational and equipment issues.*

5.13 The Impact of the Baseline on ESA-listed Species

Collectively, the stressors described above have had, and likely continue to have, lasting impacts on the ESA-listed species considered in this opinion and conference report. 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 impacts (e.g., whale watching). Assessing the aggregate impacts of these stressors on the species considered in this opinion and conference report is difficult and, to our knowledge, no such analysis exists. This becomes even more difficult considering that many of the species in this opinion and conference report 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 and conference report are experiencing increases in population abundance, some are declining, and for others, their status remains unknown. Taken together, this indicates that the *Environmental Baseline* is impacting species in different ways. The species experiencing increasing population abundances 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, species may experience phenomena such as demographic stochasticity, inbreeding depression, and Allee effects, among others, that cause their limited population size to become a threat in and of itself. A thorough review of the status and trends of each species is discussed in the *Status of Listed Resources* section of this opinion and conference report.

6 EFFECTS OF THE ACTION ON ESA-LISTED SPECIES AND CRITICAL HABITAT

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

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

As the statutory definitions are currently applied, a wide range of behavioral reactions may qualify as Level B harassment under the MMPA or harassment pursuant to the ESA, including but not limited to avoidance of the sound source, temporary changes in vocalizations or dive patters, temporary avoidance of an area, or temporary disruption of feeding, migrating, or reproductive behaviors. The U.S. Navy modeled estimates calculated using the risk continuum function (described further in Section 3.8.1) 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 this consultation, we interpret "harass" to mean an intentional or negligent action that has the potential to injure an animal or disrupt its normal behaviors to a point where such behaviors are abandoned or significantly altered.[9]

The jeopardy analysis relies upon the regulatory definition of "to jeopardize the continued existence of a listed species," which is "to engage in an action that would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of an ESA-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.

In this section, we identify the potential stressors that are likely to have direct and indirect effect on the physical, chemical, and biotic environment of the action area. Then, we present our risk assessment regarding the potential for exposure and effects from exposure to SURTASS LFA sonar. We then describe the mitigation proposed to reduce the likelihood of exposure to listed species from those stressors. Our analysis assumes that these stressors pose no risk to ESA-listed species if these stressors do not co-occur with those species in space or time. We recognize that

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[9] NMFS has not adopted a regulatory definition of harassment under the ESA. The World English Dictionary defines harass as "to trouble, torment, or confuse by continual persistent attacks, questions, etc." The U.S. Fish and Wildlife Service defines "harass" in its regulations as "an intentional or negligent act or omission which creates the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavioral patterns which include, but are not limited to, breeding, feeding, or sheltering (50 CFR 17.3). The interpretation we adopt in this consultation is consistent with our understanding of the dictionary definition of harass and is consistent with the Service's interpretation of the term.

the sonar could have indirect, adverse effects on listed species by disrupting marine food chains, a species' predators, or a species' competitors; however, we did not identify situations where this concern might apply to species under NMFS' jurisdiction. We conclude by examining the likely responses of ESA-listed species given exposure to SURTASS LFA sonar.

6.1 Stressors Associated with the Proposed Action

The primary stressors associated with the SURTASS LFA sonar operations are:

(1) The ships and ship traffic associated with the proposed sonar activities;

(2) The passive sonar system (SURTASS);

(3) The low-frequency active (LFA sonar) component of the sonar system; and

(4) The high-frequency (HF/M3) active component of the monitoring/mitigation system.

During consultation, we made the final ESA determinations of effect for each of these stressors. Stressors that are not likely to adversely affect ESA-listed species or designated critical habitat are discussed in Section 6.1.1. In this section, we also summarize our conclusions for these stressors. Stressors that are likely to adversely affect ESA-listed species or designated critical habitat are discussed briefly in Section 6.1.2, and then carried forward in the exposure and response analysis.

6.1.1 Stressors that are Not Likely to Adversely Affect ESA-listed Resources

We determined that the following stressors are not likely to adversely affect ESA-listed species or designated critical habitat.

6.1.1.1 Vessel Operation – Noise

Sounds emitted by large vessels (such as SURTASS LFA vessels) can be characterized as low-frequency, continuous, or tonal, and sound pressure levels at a source will vary according to speed, burden, capacity and length (Kipple and Gabriele 2007; McKenna et al. 2012; Richardson et al. 1995b).

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 2000b; Bain et al. 2006; Bauer 1986; Bejder et al. 1999; Bejder and Lusseau. 2008; Bejder et al. 2009; Bryant et al. 1984; Corkeron 1995; Erbe 2002b; Felix 2001; Goodwin and Cotton 2004; Lemon et al. 2006; Lusseau 2003; Lusseau 2006; Magalhaes et al. 2002; Nowacek et al. 2001; Richter et al. 2003b; Scheidat et al. 2004; Simmonds 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; Felix 2001; Goodwin and Cotton 2004; Hewitt 1985; Lusseau 2003; Lusseau 2006; Magalhaes et al. 2002; Nowacek et al. 2001; Richter et al. 2006; Richter et al. 2003b; Scheidat et al. 2004; Simmonds 2005; Watkins 1986; Williams and Ashe 2007; Williams et al. 2002b; Wursig et al. 1998) (Acevedo 1991; Aguilar Soto et al. 2006; Arcangeli and Crosti 2009; Au and Green 2000b; Christiansen et al. 2010; Erbe 2002b; Noren et al. 2009; Stensland and Berggren 2007; Stockin et al. 2008; Williams et al. 2009), 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 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; Kruse 1991; Lusseau 2003; Nowacek et al. 2001; Stensland and Berggren 2007; Williams and Ashe 2007; Williams et al. 2002b);

- *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; Hewitt 1985; Kruse 1991; Lundquist et al. 2012; Lusseau 2003; Tseng et al. 2011; Williams et al. 2002b);

- *The vessel's speed and vector* (Williams et al. 2002b);

- *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 1991; 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) (Lusseau 2003; Lusseau 2006; Polagye et al. 2011; Williams et al. 2002b);

- *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* (Lusseau 2003; Lusseau 2006; Williams et al. 2002b; 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 1991). Some individuals also dove and remained motionless, waiting until the vessel moved past their location. Most animals finding themselves in confined spaces, such as shallow bays, during vessel approaches tended to move towards more open, deeper waters (Kruse 1991). We assume that this movement would give them greater opportunities to avoid or evade vessels as conditions warranted.

Although most of these studies focused on small cetaceans (for example, bottlenose dolphins, spinner dolphins, [*Stenella longirostris*] spotted dolphins, [*Stenella* spp.] harbor porpoises, beluga whales, [*Delphinapterus leucas*] and killer whales), studies of large whales have reported similar results for fin and sperm whales (Williams et al. 2002b). Baker et al. (1983) reported that humpbacks in Hawaii responded to vessels at distances of two to four km (1.1 to 2.2 nmi). Richardson et al. (1985a) reported that bowhead whales (*Balaena mysticetus*) swam in the opposite direction of approaching seismic vessels at distances between one and four km (0.5 to 2.2 nmi) 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 U.S. 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.

We would expect similar behavioral avoidance responses, if any, from ESA-listed sea turtles and fish. Hazel et al. (2007) documented green sea turtle avoidance responses to oncoming vessels. Scalloped hammerhead sharks and sturgeon are highly mobile species and, if present in proximity to a SURTASS LFA sonar vessel, would be expected to temporarily move from the immediate area.

We recognize that SURTASS LFA sonar vessels almost certainly incorporate quieting technologies that reduce their acoustic signature (relative to the acoustic signature of similarly sized vessels) in order to reduce their vulnerability to detection by enemy vessels (Southall 2005). Additionally, the low ship speeds of SURTASS LFA sonar vessels result in very little engine or propeller cavitation noise being generated into the surrounding marine environment (Navy 2015b). Nevertheless, we do not assume that any quieting technology or low travel speeds would be sufficient to prevent marine animals from detecting sounds produced by approaching U.S. Navy vessels and perceiving those sounds as predatory stimuli. We also consider evidence that factors other than received sound level, including the activity state of animals exposed to different sounds, the nature and novelty of a sound, and spatial relations between sound source and receiving animals (i.e., the exposure context) strongly affect the probability of a behavioral response (Ellison et al. 2012a).

We considered the research and reports cited above and conclude that ESA-listed marine mammals, sea turtles, and fish are likely to either not react or exhibit an avoidance behavior. Most avoidance responses would consist of slow movements away from vessels the animals perceive are on an approaching course, perhaps accompanied by slightly longer dives in marine mammals and turtles. Most of the changes in behavior would consist of a temporary shift from behavioral states that have low energy requirements (resting or milling) to behavioral states with higher energy requirements (active swimming or traveling) and then returning to the resting or milling behavior shortly thereafter.

Behavioral disruptions of ESA-listed species resulting from the presence of vessels or submarines are expected to be temporary. Animals are expected to resume their migration, feeding, or other behaviors with no significant impact to their breeding, feeding or sheltering. Marine mammals react to vessels in a variety of ways and seem to be generally influenced by the activity the marine mammal is engaged in when a vessel approaches (Richardson et al. 1995d).

Some respond negatively by retreating or engaging in antagonistic responses while other animals ignore the stimulus altogether (Terhune and Verboom 1999; Watkins 1986). Given the short duration of vessel noise stressors, the infrequency of this stressor, and the temporary nature of biological responses of marine mammals, sea turtles, and fish to this stressor, these ESA-listed species are either not likely to respond or are not likely to respond in ways that might be adverse (the responses might represent an approach or attentive movement, a small change in orientation in the waters, etc.).

In conclusion, ESA-listed species are either not likely to respond to vessel noise or are not likely to measurably respond in ways that would significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding or sheltering. Therefore, the effects of noise from U.S. Navy vessels on ESA-listed species is considered insignificant and is not likely to adversely affect ESA-listed species. Vessel noise will not be considered further in this opinion and conference report.

6.1.1.2 Vessel Operation - Discharges

Operation of the SURTASS LFA sonar vessels will result in discharges incidental to normal operations of a vessel. The International Convention for the Prevention of Pollution from Ships (MARPOL73/78) prohibits certain discharges of oil, noxious liquid substances, sewage, garbage, and air pollution from vessels. The Convention is implemented by the Act to Prevent Pollution from Ships (APPS; 33 U.S.C. §§ 1905-1915 et seq.), which establishes requirements for the operation of U.S. Navy vessels. The SURTASS LFA sonar vessels will operate in compliance with these requirements. Despite the precautionary measures taken by the U.S. Navy to ensure such incidents do not occur, it is possible for an unintentional discharge of pollutants to occur. Therefore, we also evaluated the potential for a small pollutant discharge that could occur during project activities to impact ESA-listed species. However, because of the open ocean environment in which the proposed action will occur, the duration and small spatial extent of such a spill, and the wide-ranging life histories and mobility of ESA-listed species that may occur in the action areaarea, the effects of a small spill are considered insignificant. As a result, discharges incidental to normal operations of a vessel are not likely to adversely affect ESA-listed species and will not be considered further in this opinion and conference report.

6.1.1.3 Vessel Operation – Ship Strike

Potential adverse effects on ESA-listed species could occur through ship strikes. During SURTASS LFA sonar operations, the combination of ship speeds and the three elements of the U.S. Navy’s monitoring and mitigation program would be expected to virtually eliminate the risk of ship strikes for cetaceans. During the ten years of MMPA incidental take authorizations for SURTASS LFA sonar vessel operations there have not been any ship strikes (Navy 2011). Therefore we determine the risk of ship strikes or disturbance from ships for cetaceans to be discountable.

ESA-listed pinnipeds, fish, and sea turtles may also be vulnerable to ship strike (e.g., (Brown and Murphy 2010; Hazel et al. 2007). However, because of: (1) the slow speed of the SURTASS LFA sonar vessels (3 knots; 3.5 mph during sonar operation and 10 knots; 11.5 kph during transit) and the ability of these species to move out of the way of a such vessels; (2) the low densities of these species in the offshore areas where the vessels would be maneuvering; and (3) the U.S. Navy's monitoring and mitigation program, we believe the risk of ship strikes for pinnipeds, fish, and sea turtles to be discountable. In addition to the slow speed of travel, the design of the T-AGOS vessels, with the catamaran-type split, hull shape and enclose propeller system, make the potential for ship strike unlikely. Therefore, endangered or threatened marine mammals, sea turtles, and fish are not likely to be struck by the USNS *ABLE*, USNS *EFFECTIVE*, USNS *IMPECCABLE*, or USNS *VICTORIOUS* while these ships are underway and this potential stressor is not discussed further in this opinion and conference report.

6.1.1.4 Passive System Component - SURTASS

The passive, or listening, part of the SURTASS LFA sonar system is the Surveillance Towed Array Sensor System or "SURTASS." This system detects returning echoes from submerged objects, such as threat submarines, through the use of hydrophones. Part of the SURTASS transforms mechanical energy (received acoustic sound wave) to an electrical signal that can be analyzed by the processing system of the sonar. The SURTASS hydrophones are mounted on a receive array (horizontal line array [HLA]) that is towed astern of the vessel. The SURTASS LFA sonar vessel tows the hydrophone array at a minimum speed of 3 knots (3.5 mph) through the water to maintain the proper towed array geometry for maximum sonar system performance. The return signals, which are usually below background or ambient noise level, are then processed and evaluated to identify and classify potential underwater threats.

Because this sonar system does not transmit energy into the marine environment and because this system does not pose substantial strike risk over that of the towing vessel, we conclude that it is not likely to adversely affect ESA-listed species or designated critical habitat in the action area. Therefore, the effect of this potential stressor is insignificant and will not be discussed further in this opinion and conference report.

6.1.1.5 High Frequency (HF/M3) Sonar

As a mitigation measure to reduce the exposure of marine mammals and sea turtles to SURTASS LFA sonar, the U.S. Navy will operate the HF/M3 sonar continuously during the course of active sonar transmissions. The HF/M3 sonar, which is a U.S. Navy-developed, enhanced HF commercial sonar, designed to detect, locate, and track marine mammals and, to some extent, sea turtles and large fish (e.g., sturgeon and sharks), that may pass close enough to the SURTASS LFA sonar's transmitter array to enter the LFA sonar mitigation and buffer zones. Analysis and testing of the HF/M3 sonar operating capabilities indicate that the system substantially increases the probability of detecting marine mammals within the LFA sonar mitigation and buffer zones. The HF/M3 sonar provides an excellent monitoring capability (particularly for medium to large marine mammals) beyond the LFA sonar mitigation and buffer zones, out to 2 to 2.5 km (1.08 to 1.35 nmi).

The source level required for the HF/M3 sonar to effectively detect marine mammals (and possibly sea turtles) out to the 180-dB LFA sonar mitigation and buffer zones under the most adverse oceanographic conditions (low echo return and high ambient noise) is on the order of 220 dB re: 1 µPa (rms) at 1 m. The U.S. Navy designed the HF/M3 sonar to be as benign as possible within the marine environment to minimize potential effects to marine mammals and sea turtles. These features include:

1. The HF/M3 sonar source frequency is greater than 30 kHz, which pushes its frequency band well away from the best hearing bandwidth of mysticetes, pinnipeds, and sea turtles, but within the best hearing bandwidth of odontocetes;
2. A duty cycle that is variable, but below ten percent;
3. A maximum HF/M3 sonar pulse with a duration of 40 msec. Ridgway et al. (1997) and Schlundt et al. (2000b) reported that measured TTS in bottlenose dolphins for a 20 kHz, one-second pulse occurred at response levels of 193 to 196 dB. For a 30 kHz, 40-msec pulse, the estimated range from the HF/M3 sonar of 193 dB response level would be 22 m (72 ft);
4. A transmission loss that is very high because of the high frequency of the sound source.

In addition, as supplementary safety measures, HF/M3 sonar source level would be ramped up over a five-minute period to alert a marine mammal that was close to the sonar and provide it time to move away from the sound source.

The application of these operational procedures reduces potential impacts of the HF/M3 sonar on marine mammals to negligible (i.e., insignificant) levels. Therefore the potential effect of HF/M3 sonar on ESA-listed species is considered insignificant, is not likely to adversely affect ESA-listed species, and will not be considered further in this opinion and conference report.

6.1.2 Stressors that are Likely to Adversely Affect ESA-listed Resources

It has been determined that the following stressor is likely to adversely affect ESA-listed resources.

6.1.2.1 Active System Component - LFA sonar

The LFA system consists of up to 18 low-frequency acoustic-transmitting projectors that are suspended from a cable beneath the ship. The source level of an individual projector in the LFA sonar array is about 215 dB re: 1 µPa (rms) at 1 m, and the sound field of the array can never have a sound pressure level higher than that of an individual projector. The typical LFA sonar signal is not a constant tone, but is a transmission of various signal types that vary in frequency and duration (including continuous wave and frequency-modulated signals). The U.S. Navy refers to a complete sequence of sound transmissions as a “ping” which can range from between six and 100 seconds, with no more than ten seconds at any single frequency. The time between pings will typically range from six to 15 minutes. The U.S. Navy can control the average duty cycle (the ratio of sound “on” time to total time) for the system but the duty cycle cannot exceed 20 percent; based on operations since 2003, the duty cycle has averaged about 7.5 percent.

The LFA sonar mitigation zone covers a volume ensonified to a received level at or above 180 dB re: 1 µPa (rms) by the SURTASS LFA sonar transmit array. Based on spherical spreading, this zone will vary between the nominal ranges of 0.75 to one km (0.40 to 0.54 nmi) from the source array ranging over a depth of approximately 87 to 157 m (285 to 515 ft). The center of the array is at an approximate depth of 122 m (400 ft).

We assume that the propagation of signals transmitted from LFA sonar systems would be affected by surface ducts, sound channels, convergence zones, and bottom interactions. For more complete discussion of sound propagation in marine environments, readers should refer to Richardson et al. (1995d), Appendix B of the U.S. Navy’s EIS on SURTASS LFA sonar (Navy 2001c), and the Effects of the Action section of NMFS’ May 30, 2002 (NMFS 2002) and August 12, 2007 (NMFS 2007b) final rules, and biological opinions on SURTASS LFA sonar.

Sections 6.3 and 6.5 present our exposure and response analyses for this stressor.

6.2 Risk Assessment Framework – Marine Mammals

For marine mammals that are exposed to SURTASS LFA sonar transmission, this section examines the relationship between received levels and ecological effects; the plausibility of adverse effects occurring as a result of exposure; and linkages between measurable ecological effects and vital rates or biologically-important behavior in populations of ESA-listed marine mammals, which are the endpoints for this assessment. In this analysis we consider potential for non-auditory injury, auditory injury (PTS), TTS, behavioral changes, masking, and stranding as a result of exposure to SURTASS LFA sonar.

To assess the plausibility of adverse effects resulting from exposure to SURTASS LFA sonar, we evaluated various lines of evidence from published and unpublished sources on the effects of SURTASS LFA sonar on threatened and endangered species, other studies of low frequency sound, and reports on the effects of other sonars. We evaluate the same lines of evidence to establish linkages between the potential effects of SURTASS LFA sonar on biologically important behavior of ESA-listed species exposed to the sonar. We also consider information provided in annual and summary reports on previous SURTASS LFA sonar activities.

6.2.1 Non-Auditory Injury

Potential non-auditory injury to marine mammals from LFA 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) would not occur from exposure to LFA 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, sonar induced bubble formation, and nitrogen decompression 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.

6.2.1.1 Acoustic Resonance

Physical effects, such as direct acoustic trauma or acoustically enhanced bubble growth, require relatively intense received energy that would only occur at short distances from high-powered sonar sources (Nowacek et al. 2007; Zimmer and Tyack 2007). The best available scientific information shows that, while resonance can occur in marine animals, this resonance does not necessarily cause injury, and any such injury is not expected to occur below a received level of 180 dB SPL. Damage to the lungs and large sinus cavities of cetaceans from air space resonance is not likely to occur because resonance frequencies of marine mammal lungs are below that of the LFA sonar signal (Finneran 2003). Further, biological tissues are heavily damped and tissue displacement at resonance is predicted to be exceedingly small. In addition, lung tissue damage is generally uncommon in acoustic-related strandings (Southall et al. 2007a)

6.2.1.2 Bubble Formation

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. 2001c). 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. 2008b). 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).

Despite the increase in research and literature, there remains scientific disagreement and a lack of scientific data regarding the evidence for gas bubble formation as a causal mechanism between certain types of acoustic exposures and stranding events in marine mammals. These issues include: (1) received acoustic exposure conditions; (2) pathological interpretation; (3) acoustic exposure conditions required to directly induce physiological trauma; (4) behavioral reactions caused by sound exposure such as atypical dive patterns; and (5) the extent of postmortem artifacts (Southall et al. 2007a).

The hypotheses for gas bubble formation related to beaked whale strandings is that beaked whales potentially have strong avoidance responses to MFA sonars because they sound similar to their main predator, the killer whale (Baird et al. 2008b; Cox et al. 2006; Hooker et al. 2009b; Southall et al. 2007a; Zimmer and Tyack 2007). Because SURTASS LFA sonar transmissions are lower in frequency (less than 500 Hz) and dissimilar in characteristics from those of marine mammal predators, the above scientific studies do not provide additional evidence that SURTASS LFA sonar has caused behavioral reactions, specifically avoidance responses, in beaked whales. Thus, SURTASS LFA sonar transmissions are not expected to cause gas bubble formation or beaked whale strandings.

6.2.1.3 Nitrogen Decompression

Although not a direct injury, variations in marine mammal diving behavior or avoidance responses could possibly result in nitrogen tissue supersaturation and nitrogen off-gassing. Nitrogen supersaturation and off-gassing levels could result in deleterious vascular and tissue bubble formation (Hooker et al. 2012; Jepson et al. 2003; Saunders et al. 2008a). 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. 2008b). 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. Fahlman et al. (2009) suggested that repeated dives might result in tissue and blood levels of nitrogen sufficient to cause symptomatic 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 MFA sonar (Fernandez et al. 2005a; Jepson et al. 2003) could stem from a behavioral response that involves repeated dives, shallower than the depth at which lung collapse would occur. A bottlenose dolphin was trained to repetitively dive to elevate nitrogen saturation to the point that asymptomatic nitrogen bubble formation was predicted to occur. However, inspection of the vascular system of the dolphin via ultrasound did not demonstrate the formation of any nitrogen gas bubbles (Houser 2010).

More recently, modeling has suggested that the long, deep dives performed regularly by beaked whales over a lifetime could result in the saturation of tissues (e.g., fat, bone lipid) to the point that they are supersaturated when the animals are at the surface (Hooker et al. 2009a; Saunders et al. 2008b). Proposed adaptations for prevention of bubble formation under conditions of persistent tissue saturation have been suggested (Fahlman et al. 2006; Hooker et al. 2009a). 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.

6.2.2 Hearing Loss – PTS and TTS

The most familiar effect of exposure to high intensity sound is hearing loss, meaning an increase in the hearing threshold. The meaning of the term “hearing loss” does not equate to “deafness.” This phenomenon is called 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. TTS may last from minutes to days. The following physiological mechanisms may result in TTS:

1. Reduced sensitivity of the sensory hair cells in the inner ear as a result of their being over-stimulated;
2. Modification of the chemical environment within sensory cells;
3. Displacement of certain inner ear membranes;
4. Increased blood flow; and

5. Post-stimulation reduction in both efferent (impulses traveling from the central nervous system to the peripheral sensory tissue) and sensory output (Kryter 1994; Southall et al. 2007a).

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, it is likely that 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.

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. Figure 10 shows one hypothetical threshold shift that completely recovers, a TTS, and one that does not completely recover, leaving some PTS. The actual amount of threshold shift depends on the amplitude, duration, frequency, temporal pattern of the sound exposure, and on the susceptibility of the individual animal.

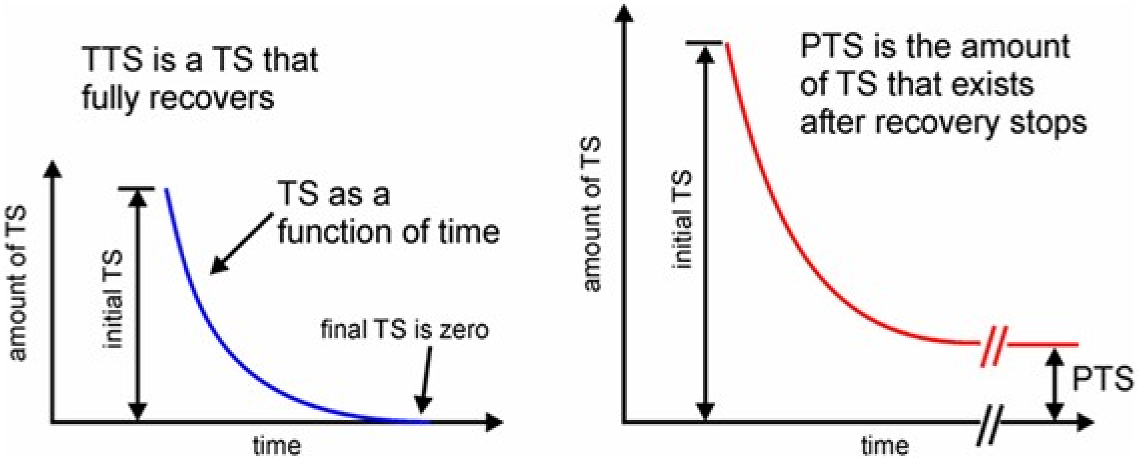

Figure 10. Two Hypothetical Threshold Shifts, Temporary and Permanent.

Both auditory trauma and auditory fatigue may result in hearing loss. Many are familiar with hearing protection devices (i.e., ear plugs) required in many occupational settings where persistent 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 “temporary threshold shift;” however, we use 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). The actual amount of threshold shift depends on the amplitude, duration, frequency, and temporal pattern of the sound exposure.

Hearing loss in marine mammals has been studied by a number of investigators. (Finneran et al. 2005; Finneran and Schlundt 2010; Finneran et al. 2007; Finneran et al. 2000b; Finneran et al. 2002b; Lucke et al. 2009; Mooney et al. 2009a; Mooney et al. 2009b; Nachtigall et al. 2003; Schlundt et al. 2000a). The studies of marine mammal auditory fatigue were all designed to determine relationships between TTS and exposure parameters such as level, duration, and frequency.

In these studies, hearing thresholds were measured in trained marine mammals before and after exposure to intense sounds. The difference between the pre-exposure and post-exposure thresholds indicated the amount of TTS. Species studied include the bottlenose dolphin (total of nine individuals), beluga (2), harbor porpoise (1), finless porpoise (2), California sea lion (3), harbor seal (1), and Northern elephant seal (1). Some of the more important data obtained from these studies are onset-TTS levels— exposure levels sufficient to cause a just-measurable amount of TTS, often defined as six dB of TTS (Schlundt et al. 2000a). These criteria for onset-TTS are very conservative, and it is not clear that this level of threshold shift would have a functional effect on the hearing of a marine mammal in the ocean.

Some of 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).
- 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. (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).
- The maximum TTS after tonal exposures occurs one-half to one octave above the exposure frequency (Finneran et al. 2007; Schlundt et al. 2000a). TTS from tonal exposures can thus extend over a large (greater than one octave) frequency range.
- For bottlenose dolphins, sounds with frequencies above ten kHz are more hazardous than those at lower frequencies (i.e., lower sound exposure levels required to affect hearing) (Finneran and Schlundt 2010).
- The amount of observed TTS tends to decrease with increasing time following the exposure; however, the relationship is not monotonic. The amount of 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) 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.

A study of TTS in harbor porpoises used a seismic airgun as a stimulus (Lucke et al. 2009). Airguns produce an impulsive signal and have a broad frequency range but also have substantial energy in the low frequency region. A small airgun was used in proximity to the animals (between 14 to 150 m [46 to 492 ft]), a context that is likely to enhance behavioral responsiveness. The harbor porpoises showed a behavioral response at a received level of 174 dB re: 1 µPa (peak-to-peak), which is equivalent to a sound exposure level of 145 dB re: 1 μPa^2-sec (Lucke et al. 2009). Harbor porpoise hearing was tested at a frequency of 4 kHz and TTS was detected at a received level of 199.7 dB re: 1 µPa (peak-to-peak), which is equivalent to a sound exposure level of 164.3 dB re: 1 μPa^2-sec (Lucke et al. 2009). These data are intriguing and clearly indicate a need for additional research. Unfortunately, only one individual was tested in this study. The applicability of these results to SURTASS LFA sonar is uncertain, given the large differences in source characteristics between airguns and LFA sonar. Furthermore, LFA sonar typically operates in water deeper and further offshore than most harbor porpoise habitats. Indeed, harbor porpoises are found in only one of the SURTASS LFA sonar operating areas analyzed, for which zero exposures at levels greater than 180 dB sound pressure level were found. Nevertheless, this study indicates that further study of TTS in porpoises is warranted. Ideally, additional harbor porpoise individuals as well as additional high-frequency hearing species would be tested. If this type of results are confirmed for harbor porpoise or found in other HF hearing species, then the analyses for those species would merit revision.

In a study on the effects of noise level and duration of TTS in a bottlenose dolphin, Mooney et al. (2009a) exposed a bottlenose dolphin to octave-band noise (four to eight kHz) of varying durations (two to 30 minutes) and sound pressure level received levels (130 to 178 dB re: 1 μPa). The results of the Mooney et al. study indicated that shorter-duration sound exposures often require greater sound energy to induce TTS than longer-duration exposures and also supported the trend that longer-duration exposures often induce greater amounts of TTS, which concurrently require longer recovery times.

In a controlled exposure experiment, Mooney et al. (2009b) demonstrated that MFA sonar could induce temporary hearing loss in a bottlenose dolphin (*Tursiops truncatus*). Temporary hearing loss was induced by repeated exposure to a sound exposure level of 214 dB re: 1 μPa^2-sec. Subtle behavioral alterations were also associated with the sonar exposures. At least with one odontocete species (common bottlenose dolphin), sonar can induce both TTS and mild behavioral effects; but exposures must be prolonged with high exposure levels to generate these effects. The received level used in the Mooney et al. (2009b) experiment was a sound pressure level of 203 dB, which equates to the received level approximately 40 m (131 ft) from an MFA sonar operated at a sound pressure level of 235 dB (source level). Mooney et al. (2009b) concluded that in order to receive a sound exposure level of near 214 dB, an animal would have to remain in proximity of the moving sonar, which is transmitting for 0.5 seconds every 24 seconds over an approximately two to 2.5 minute period, an unlikely situation.

Sound exposure levels necessary for TTS onset for pinnipeds in water have been measured for harbor seals, California sea lions, and northern elephant seals. As reported by Southall et al. (2007a), Kastak et al. (2005) presented comparative analysis of underwater TTS for pinnipeds. This indicated that in harbor seals, a TTS of approximately six dB occurred with a 25-minute exposure to 2.5 kHz octave-band noise of 152 dB sound pressure level (183 dB sound exposure level); a California sea lion showed TTS-onset under the same conditions at 174 dB sound pressure level (206 dB sound exposure level); and a northern elephant seal under the same conditions experienced TTS-onset at 172 dB sound pressure level (204 dB sound exposure level). Finneran et al. (2003) exposed two California sea lions to single underwater pulses from an arc-gap transducer and found no measurable TTS following exposures of up to 183 dB sound pressure level (215 dB sound exposure level).

Animals suffering from TTS over longer periods of time, such as days, may be considered to have a change in a biologically significant behavior, as they may be prevented from detecting sounds that are biologically relevant, including communication sounds, sounds of prey, or sounds of predators. As noted by Mooney et al. (2009a), shorter duration sound exposures can require greater sound energy to induce TTS than longer duration exposures, and longer duration exposures can induce greater amounts of TTS. In assessing the potential for LFA sonar transmissions to cause TTS, the much shorter length of the LFA sonar signal (typically one minute) versus the above studies (two to 30 minutes) must be considered.

In a recent synthesis of TTS experiments and information, Tougaard et al. (2014) report that although TTS by definition is considered a temporary effect, recent experiments with terrestrial mammals suggest that an animal’s auditory system may not recover fully from severe TTS and that permanent degeneration of the cochlear nerve system may result from recoverable TTS episodes and possibly repeated severe TTS can cause accumulated damage that could lead to a neurologically-based PTS. Additionally, the authors reported on the potential for earlier TTS studies to have overestimated the levels for onset of TTS due to high levels of ambient noise in the experimental setting as well as the use of older animals without optimal hearing (Tougaard et al. 2014). While other recent research has shown that harbor and finless porpoises are more sensitive to sound than expected from extrapolations of research results on bottlenose dolphins (Schlundt et al. 2000a), Tougaard et al. (2014) caution that it is not known whether these results were due to differences in experimental methodology, masking levels, or the age and health of the subject animals, but the reliability of extrapolating TTS thresholds between species relies on careful examination of such factors.

TTS studies on harbor porpoises have indicated a higher susceptibility to noise than other odontocetes and more sensitivity to sound than expected from extrapolations based on results from bottlenose dolphins and belugas (Kastelein et al. 2012; Kastelein et al. 2014a; Lucke et al. 2009). Kastelein et al. (2014a) induced up to about 30 dB re 1-μPa^2-sec (SEL) of TTS by exposure of an eight-year old harbor porpoise to frequency modulated sweeps between one andtwo kHz (without harmonics) in various combinations of duration (1.9 to 240 minutes), intensity (144 to 179 dB re: 1 μPa SPL), and duty cycle (five to 100 percent), with SEL thresholds for TTS occurring between 188 and 196 dB re: 1 μPa^2-sec. Additionally, Kastelein et al. (2014a) found that the inter-pulse interval of the experimental sound was an important parameter in determining the magnitude of noise-induced TTS.

In a separate study, Kastelein et al. (2014b) determined the effects on the hearing of a seven-year old harbor porpoise after exposure to a 6.5 kHz continuous signal for 60 minutes at received SPLs ranging from 118 to 154 dB re: 1 μPa. The authors determined that the hearing frequency at which the maximum TTS occurred for the harbor porpoise was dependent on the noise level

(SPL); the higher the SPL, the higher the TTS induced at frequencies higher than the exposure frequency (below 148 dB re: 1μPa, the maximum TTS was at 6.5 kHz, whereas above 148 dB re: 1 μPa, the maximum TTS was at 9.2 kHz) (Kastelein et al. 2014b). Last, Kastelein et al. (2014b) noted that recovery from TTS in the harbor porpoise, after small reductions in hearing sensitivity (threshold shifts less than 15 dB), is relatively quick (within 60 minutes) and that such a short-term reduction in hearing likely has little impact but repeated hearing reductions over hours or days may become ecologically significant for the harbor porpoise. While noteworthy because this study induced TTS at received levels lower than 180 dB, no individual marine mammal will be exposed to a continuous acoustic signal for 60 minutes as a result of the proposed action.

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 growth relationships from marine and terrestrial mammals to determine the exposure levels capable of producing this amount of TTS.

Since the boundary line between TTS and PTS is neither clear, definitive, nor predictable for marine mammals, NMFS adopted the standard that 20 dB of threshold shift defines the onset of PTS (i.e., a shift of 20 dB in hearing threshold) (NMFS 2002). NMFS used this same standard in the 2007 SURTASS LFA Final Rule (72 FR 46846). Southall et al. (2007b) proposed injury criteria for individual LF/MF/HF marine mammals exposed to non-pulsed sound types, which included discrete acoustic exposures from SURTASS LFA sonar. The proposed injury criteria for cetaceans and pinnipeds in water are sound exposure levels of 215 dB received level and 203 dB received level, respectively. An 18-dB adjustment must be made for the longer LFA sonar signal (nominally 60 seconds) resulting in injury criteria for SURTASS LFA sonar for LF/MF/HF cetaceans of a sound exposure level of 197 dB received level and for pinnipeds in water an sound exposure level of 185 dB received level. The SURTASS LFA sonar injury criterion for all marine mammals was a sound pressure level of 180 dB received level (Navy 2001b; Navy 2007; Navy 2012a), which is noticeably lower and, therefore, more conservative, than the injury criteria proposed by Southall et al. (2007b).

The center of LFA sonar arrays would operate at a depth of approximately122 m (400 ft), which would generally place the array and the main sound transmission beam produced by the arrays below the mixed-layer depth of an ocean. The transmitted signal would move in an extended wave through the ocean, initially moving away from the surface, then rising at some distance from the source, before sinking again (in some instances affected by ducting, sound channels, convergence zones, and bottom interactions). At the source, an LFA sonar signal is approximately 215 dB re: 1 μPa (rms) at 1 m for a single projector. This signal would attenuate with time and distance from the source. Received levels from the array attenuate to 180 dB re: 1 µPa (rms) SPL nominally from 750 to 1,000 m (2,460 to 3,281 ft) from the LFA sonar transmit array.

The SURTASS LFA sonar has the greatest potential for injuring threatened or endangered species if an animal is close to one or more of the projectors during a ping. If that occurred, the animal could be exposed to sound pressure levels between 215 and 180 dB re: 1 µPa (rms) SPL, which could injure listed species by causing permanent threshold shifts. As described in Section 6.4.4, the U.S. Navy proposes to use HF/M3 sonar, visual monitoring, and passive acoustic monitoring, to detect animals within one to two kms of the projectors, minimizing the likelihood of this occurring. If any of these monitoring methods detects animals within this zone, the projectors would be shut down until the animal(s) moved out of the zone.

Testing of the HF/M3 sonar has demonstrated a probability of single-ping detection above 95 percent within the LFA sonar mitigation and buffer zones for many marine mammals (Ellison and Stein 1999). Under normal operating conditions, marine mammals will receive multiple HF/M3 pings, increasing the probability of detection within the LFA mitigation zone to near 100 percent (DON 2001). Combined with the visual monitoring and passive acoustic monitoring protocols, the risk of marine mammals being exposed to sound pressure levels in excess of 180 dB re: 1 µPa (rms) approaches zero.

Therefore, unless an animal occurred within the 180-dB re: 1 µPa (rms) LFA sonar mitigation and buffer zones during a ping, it is not likely to experience physical injury, including damage to tissues. Because of the mitigation measures, a marine mammal’s likelihood of being exposed to these sound pressure levels is very low. It is unlikely that SURTASS LFA sonar will cause TTS at a SPL of 180 dB RL in the ESA-listed marine mammals considered in this opinion and conference report and very few animals are likely at all to be affected by TTS. Additionally, the implementation of the mitigation monitoring that is a condition of SURTASS LFA sonar use would allow little potential for a marine mammal to incur TTS.

6.2.3 Behavioral Responses

The primary potential deleterious effect from SURTASS LFA sonar is change in a biologically significant behavior. The National Research Council (NRC 2005b) discussed biologically significant behaviors and possible effects and stated that an action or activity becomes biologically significant to an individual animal when it affects the ability of the animal to grow, survive, and reproduce. These are the effects on individuals that can have population-level consequences and affect the viability of the species (NRC 2005b).

The behavioral 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) (Wartzok et al. 2003). For marine mammals, a review of responses to anthropogenic sound was first conducted by Richardson and others (Richardson et al. 1995d). More recent reviews (Ellison et al. 2012a; Nowacek et al. 2007; Southall et al. 2009; Southall et al. 2007a) address studies conducted since 1995 and focus on observations where the received sound level of the exposed marine mammal(s) was known or could be estimated.

Except for some vocalization changes that may be compensating for concurrent 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. 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 level 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. 2007a). 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.

Additional discussion on potential behavioral reactions of ESA-listed marine species to U.S. Navy mid-frequency active sonar is available in the most recent NMFS biological opinions for U.S. Navy training and testing in the Hawaii-Southern California Training and Testing (HSTT) and the Mariana Islands Training and Testing (MITT) study areas (NMFS 2015a; NMFS 2015b). However, due to the differences between mid- and low-frequency active sonar, we primarily rely on the literature described below. This literature primarily addresses potential behavioral responses of marine mammals to low-frequency active sonar.

6.2.3.1 The Low Frequency Sound Scientific Research Program

The Low Frequency Sound Scientific Research Program (LFS SRP) in 1997 to 1998 provided important results on, and insights into, the types of responses of baleen whales to LFA sonar signals and how those responses scaled relative to received level and context. The results of the LFS SRP confirmed that some portion of the total number of whales exposed to LFA sonar responded behaviorally by changing their vocal activity, moving away from the source vessel, or both; but the responses were short-lived (Clark and Fristrup 2001).

In the LFS SRP LFA sonar playback experiment (Phase II), migrating gray whales avoided exposure to LFA sonar signals (source levels of 170 and 178 dB sound pressure level) when the source was placed in the center of their migration corridor. Responses were similar for the 170-dB source level LFA sonar stimuli and for the 170-dB source level 1/3rd-octave, band-limited noise with timing and frequency band similar to the LFA sonar stimulus. However, during the LFA sonar playback experiments, in all cases, whales resumed their normal activities within tens of minutes after the initial exposure to the LFA sonar signal (Clark and Fristrup 2001). Essentially, the whales made minor course changes to go around the source. When the source was relocated within the outer portion of the migration corridor (twice the distance offshore), and the source level was increased to reproduce the same sound field for the central corridor playback condition, the gray whales showed little to no response to the LFA sonar source. This result stresses the importance of context in interpreting the animals' behavioral responses to underwater sounds and demonstrates that received level is not necessarily a good predictor of behavioral impact.

The Phase II research also demonstrated that it may not be valid to generalize results from inshore studies (2 km [1.1 nmi] from shore when 50 percent of the whales avoided SURTASS LFA sonar stimuli at received levels of 141 +3 dB) to sources that are offshore (4 km [2.2 nmi] from shore) of migrating whales (whales did not avoid offshore sources at received levels of 140 dB). This implies that the inshore avoidance model, in which 50 percent of the whales avoid exposure to levels of 141 +3 dB, may not be valid for whales in proximity to an offshore source (Buck and Tyack 2000).

The LFS SRP also conducted field tests to examine the effects of LFA sonar transmissions on foraging fin and blue whales off San Nicolas Island, California (Phase I). Overall, whale encounter rates and dive behavior appeared to be more strongly linked to changes in prey abundance associated with oceanographic parameters rather than LFA sonar sound transmissions (Croll et al. 2001b). However, five out of six North Atlantic right whales exposed to an acoustic alarm interrupted their foraging dives, although the alarm signal was long in duration, lasting several minutes, and purposely designed to elicit a reaction from the animals as a prospective means to protect them from ship strikes (Nowacek et al. 2004a). Although the animal's received sound pressure level was similar in the latter two studies (133 to 150 dB re: 1 µPa), the frequency, duration, and temporal pattern of signal presentation were different. Additionally, the right whales did not respond to playbacks of either right whale social sounds or vessel noise, highlighting the importance of the sound characteristics, species differences, and individual sensitivity in producing a behavioral reaction.

In the final phase of the LFS SRP (Phase III), the effect of LFA sonar on humpback whales during the winter mating season was investigated. Both Miller et al. (2000a) and Fristrup et al. (2003) published results from tests conducted with male humpback singers off the Big Island, Hawaii during which they evaluated variation in song length as a function of exposure to LFA sonar sounds. Fristrup et al. (2003) used a larger data set to describe song length variability and to explain song length variation in relation to LFA sonar broadcasts. In spite of methodological and sample size differences, the results of the two analyses were generally in agreement, and both studies indicated that humpback whales might lengthen their songs in response to low frequency broadcasts. The Fristrup et al. (2003) results also provided a detailed picture of short-term response as compared to behavioral variation observed in the absence of the stimuli. These responses were relatively brief in duration, with all observed effects occurring within two hours of the last LFA sonar source transmission. It should be noted that these effects were not obvious to the acoustic observers on the scene, but were revealed by careful, complex post-test statistical analyses (Fristrup et al. 2003). Aside from the delayed responses, other measures failed to indicate cumulative effects from LFA sonar broadcasts, with song-length response being dependent solely on the most recent LFA sonar transmission, and not the immediate transmission history. The modeled seasonal factors (changes in density of whales sighted near shore) and diurnal factors (changes in surface social activities) did not show trends that could be plausibly explained by cumulative exposure. Increases in song length from early morning to afternoon were the same on days with and without LFA sonar transmissions, and the fraction of variation in song length that could be attributed to LFA sonar broadcast was small (less than ten percent). Fristrup et al. (2003) found high levels of natural variability in humpback song length and interpreted the whales' responses to LFA sonar broadcasts to indicate that exposure to LFA sonar would not impose a risk of dramatic changes in humpback whale singing behavior that would have demographic consequences.

Taken together, the data generated during the three phases of the LFS Scientific Research Program did not support the initial hypothesis that most baleen whales exposed to received levels near 140 dB would exhibit behavioral changes and avoid the area. These experiments, which exposed baleen whales to received levels ranging from 120 to about 155 dB, detected only minor, short-term behavioral responses. As documented below, the conclusions from the LFS SRP are supported by other studies of whale responses to low frequency, anthropogenic sounds.

6.2.3.2 Other literature

Richardson et al. (1995d) noted that avoidance reactions are the most obvious manifestations of disturbance in marine mammals. Richardson et al. (1985b) and Richardson (1997) used controlled playback experiments to study the response of bowhead whales in Arctic Alaska. In their studies, bowhead whales tended to avoid drill ship noise at estimated received levels of 110 to 115 dB and seismic sources at estimated received levels of 110 to 132 dB. Richardson et al. (1995d) concluded that some marine mammals would tolerate continuous sound at received levels above 120 dB re: 1 μPa for a few hours. These authors concluded that most marine mammals would avoid exposures to received levels of continuous underwater noise greater than 140 dB when source frequencies were in the animal's most sensitive hearing range.

Several authors noted that migrating whales are likely to avoid stationary sound sources by deflecting their course slightly as they approached a source (LGL and Greenridge 1987 in

Richardson et al. 1995d). A study examined responses of gray whales migrating along the California coast to various sound sources located in their migration corridor (Malme et al. 1983; Malme et al. 1984). Gray whales showed statistically significant responses to four different underwater playbacks of continuous sound at received levels of approximately 120 dB. The sources of the playbacks were typical of a drillship, semisubmersible, drilling platform, and production platform.

Bowhead and gray whales, showed clear patterns of short-term, behavioral disturbance in response to a variety of actual and simulated vessel activity and noise (Malme et al. 1983; Richardson et al. 1985a). Richardson et al. (1995c) noted that whales have been seen within a few kilometers of operating seismic vessels, although they added that any discomfort the seismic sound pulses may have caused remains unknown. Humpback whales showed similar patterns on their summering grounds (Bauer and Herman 1986) and on their wintering grounds (Bauer 1986) in response to vessel noise. Richardson et al. (1995c)argued that intermittent pulses with peak levels between 160 to 180 dB are less likely to cause discomfort than continuous sounds at the same sound pressure levels.

In the Caribbean, sperm whales avoided exposure to mid-frequency submarine sonar pulses, in the range 1,000 Hz to 10,000 Hz (IWC 2005a). Blue and fin whales have occasionally been reported in areas ensonified by airgun pulses. Systematic data on their reactions to airguns are generally lacking. Sightings by observers on seismic vessels off the United Kingdom suggest that, at times of good sightability, the number of blue, fin, sei, and humpback whales seen when airguns are shooting are similar to the numbers seen when the airguns are not shooting (Stone 1997; Stone 1998; Stone 2000; Stone 2001). However, fin and sei whale sighting rates were higher when airguns were shooting, which may be due to a tendency to remain at or near the surface at times of airgun operation (Stone 2003). The analysis of the combined data from all years indicated that baleen whales stayed farther from airguns during periods of shooting (Stone 2003). Baleen whales also altered course more often during periods of shooting and more were headed away from the vessel at these times, indicating some level of localized avoidance of seismic activity (Stone 2003).

Sperm whales reacted to military sonar, apparently from a submarine, by dispersing from social aggregations, moving away from the sound source, remaining relatively silent and becoming difficult to approach (Watkins et al. 1985c). Captive bottlenose dolphins and a white whale exhibited changes in behavior when exposed to one second pulsed sounds at frequencies similar to those emitted by multi-beam sonar that is used by geophysical surveys (Ridgway et al. 1997; Schlundt et al. 2000b), and to shorter broadband pulsed signals (Finneran et al. 2000a; Finneran et al. 2002a).

Behavioral changes oftentimes appear to be deliberate attempts to avoid a sound exposure or to avoid the location of the exposure site during subsequent tests (Finneran et al. 2000a; Schlundt et al. 2000b). Dolphins exposed to one-second intense tones exhibited short-term changes in behavior above received sound levels of 178 to 193 dB re: 1 μPa (rms) and belugas did so at received levels of 180 to 196 dB and above. Received levels necessary to elicit such reactions to shorter pulses were higher (Finneran et al. 2000a; Finneran et al. 2002a). Test animals sometimes vocalized after exposure to pulsed, mid-frequency sound from a watergun (Finneran et al. 2002b). In some instances, animals exhibited aggressive behavior toward the test apparatus (Finneran et al. 2000a; Ridgway et al. 1997).

Nowacek et al. (2004b) conducted controlled exposure experiments on North Atlantic right whales using ship noise, social sounds of conspecifics, and an alerting stimulus (frequency modulated tonal signals between 500 Hz and 4.5 kHz). Animals were tagged with acoustic sensors (D-tags) that simultaneously measured movement in three dimensions. Whales reacted strongly to alert signals at received levels of 133 to 148 dB sound pressure level, mildly to conspecific signals, and not at all to ship sounds or actual vessels. The alert stimulus caused whales to immediately cease foraging behavior and swim rapidly to the surface.

There is evidence that wild animals respond to human disturbance in the same way that they respond to predators (Beale and Monaghan 2004; Frid 2003; Frid and Dill 2002; Gill and Sutherland 2001; Romero 2004). These responses manifest themselves as stress responses (in which an animal perceives human activity as a potential threat and undergoes physiological changes to prepare for a flight or fight response or more serious physiological changes with chronic exposure to stressors), interruptions of essential behavioral or physiological events, alteration of an animal’s time budget, or some combinations of these responses (Frid and Dill 2002; Romero 2004; Sapolsky et al. 2000; Walker et al. 2005). These responses have been associated with abandonment of sites (Sutherland and Crockford 1993), reduced reproductive success (Giese 1996; Müllner et al. 2004), and the death of individual animals (Daan et al. 1996).

Brownell (2004b) reported observations of the effects of behavioral disturbance on the endangered western North Pacific gray whale population off the northeast coast of Sakhalin Island associated with seismic activities in that region. In 1997, various behavioral disturbances concomitant with seismic activities were observed including changes in swimming speed and orientation, respiration rates, and distribution offshore. Cumulative impacts of these short-term disturbances are not known. In 2001, seismic activities were conducted in the known feeding area of these whales. It was observed that whales left the feeding ground during these activities and moved to areas farther south. They only returned to the feeding ground after the seismic activities ceased days later. The potential impacts on these whales, especially mother-calf pairs and “skinny whales”, of being displaced to the south outside the normal feeding area are not known but are cause for concern. As reported previously, whales observed to be much skinnier than normal were first observed in 1999 and continue to be observed in the population but in smaller numbers.

One study of blue whales reported that when pulses from air guns were produced off Oregon, blue whales continued vocalizing at the same rate as before the pulses, suggesting that at least their vocalization behavior was undisturbed by the sound (McDonald et al. 1993).

Watkins (1986) reviewed data on the reactions of fin, humpback, right and minke whales that were exposed to continuous, broadband low-frequency shipping and industrial noise in Cape Cod Bay is informative. Watkins (1986) concluded that underwater sound was the primary cause of a reaction in these species of whales and that whales responded to acoustic stimuli in their range of hearing. Watkins also noted that whales showed the strongest reactions to sounds in the 15 Hz to 28 kHz range, although negative reactions (avoidance, interruptions in vocalizations, etc.) were generally associated with sounds that were either unexpected, too loud, suddenly louder or different, or perceived as being associated with a potential threat (such as an approaching ship on a collision course). In particular, whales seemed to react negatively when they were within 100 m (328 ft) of the source or when received levels increased suddenly in excess of 12 dB relative to ambient sounds. At other times, the whales ignored the source of the signal and all four species habituated to these sounds. Nevertheless, Watkins concluded that whales ignored most sounds in the background of ambient noise, including the sounds from distant human activities even though these sounds may have had considerable energies at frequencies well within the whale’s range of hearing. Further, Watkins (1986) noted that fin whales were initially the most sensitive of the four species of whales, followed by humpback whales; right whales were the least likely to be disturbed and generally did not react to low-amplitude engine noise. By the end of his period of study, Watkins (1986) concluded that fin and humpback whales have generally habituated to the continuous, broad-band, noise of Cape Cod Bay while right whales did not appear to change their response. This study covered a long enough period (the data covered 25 years of time) to provide some insight into possible long-term effects of low-frequency noise on whales, particularly since the four whale species would be exposed to continuous, low-frequency noise from shipping and other industrial sources. Given that whales in Cape Cod Bay reacted negatively to these continuous sources of anthropogenic sounds only under specific circumstances and, over time, habituated to these sounds (rather than abandoned the area), it seems unlikely that an additional, intermittent signal lasting from six to 100 seconds that is designed to mimic background, low frequency sound would have a greater negative effect on at least these species of whales (fin, humpback, right, and minke whales). The studies associated with the LFS Scientific Research Program suggest the same conclusions may also apply to blue, gray, and fin whales.

A study of a new low-frequency active sonar system that was being developed for use by the British Navy monitored the behavioral responses of marine mammals to the LFA sonar (Aicken et al. 2005). During these trials, fin whales, sperm whales, Sowerby’s beaked whales (*Mesoplodon bidens*), long-finned pilot whales (*Globicephala melas*), Atlantic white-sided dolphins (*Lagenorhynchus acutus*), and common bottlenose dolphins were observed and recorded vocalizing. These studies found no evidence of behavioral responses that could be attributed to the low-frequency active sonar during these trials (some of the responses that were observed may have been to the vessels used for the monitoring).

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 equivocal nature of behavioral effects and consequent difficulty in defining and predicting them.

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 (108 nmi) away. The OAWRS source appears to have affected more whales, by producing a greater response with a lower sound source level, than reported from the Phase III of the LFS SRP, even though OAWRS had a lower RL (88 to 110 dB re: 1 μPa) than the LFA signal. This strongly suggests that other acoustic characteristics may be responsible for the difference in observed behavioral responses. Risch et al. (2012) stated that due to differences in behavioral context, location, and proximity to the source, it is difficult to compare their findings directly to Phase III of the LFS SRP. These observations are consistent with the importance of considering context in predicting and observing the level and type of behavioral response to anthropogenic signals (Ellison et al. 2012a). Gong et al. (2014) reported on their reassessment of the effects from exposure to the OAWRS source on humpback whales and determined that the constant occurrence rate of humpback whale songs indicated that the low frequency OAWRS source transmissions had no effect on humpback whale song, contrary to what Risch et al. (2012) reported. Gong et al. (2014) attributed the variation in song occurrence Risch et al. (2012) described as due to natural causes other than sonar since the change in song occurrence began before the OAWRS began transmitting. Risch et al. (2014) responded to the Gong et al. (2014) assessment by stating that both their original and the Gong et al. (2014) findings may be correct, highlighting the multifactorial and context-dependent nature of behavioral responses. Risch et al. (2014) pointed out that the principal point of their original paper had been the importance of behavioral context, ambient noise, and the novelty of received sounds when assessing anthropogenic impacts on marine mammals.

Antunes et al. (2014) conducted exposure-control approach experiments using simulated European low frequency (one to two kHz) and mid-frequency (six to seven kHz) naval sonar to determine if transmission frequency had an effect on the behavioral response threshold for long-finned pilot whales. The experiment used a new method to quantify the dose threshold at which free-ranging long-finned pilot whales began to avoid an approaching vessel transmitting sonar, consisting of a statistical analysis of the movements of the whales to indicate changes in behavior and a Bayesian dose-response model using the observed response thresholds (Antunes et al., 2014). Pilot whale movements during the duration of sonar transmissions changed to indicate an avoidance response, with a shorter response duration than reported for other marine mammal species, including Cuvier's (*Ziphius cavirostris*) and Blainville's beaked whales

(*Mesoplodon densirostris*) and killer whales. The observed response thresholds occurred at 50 percent higher levels (approximately 170 dB re: 1 μPa or 173 dB SELcum) than described for other cetacean species (Antunes et al. 2014).

Miller et al. (2014b) conducted eight controlled exposure experiments on killer whale groups using the same simulated European low frequency (one to two kHz) and mid-frequency (six to seven kHz) naval sonar used in the Antunes et al. (2014) experiments. The SL and distance to the source were increased during each exposure to reveal response thresholds. Changes in movements (swim speed and direction) during exposure to the transmitted sonar signals were evaluated as avoidance responses and animals began to avoid the sonar at RLs SPL (142 +/- 15 dB re: 1 μPa) (Miller et al. 2014b), which is below the threshold assumed by the U.S. Navy (Finneran and Jenkins 2012). High levels of between and within individual variability were identified, and the context of exposure was recognized as an important factor, particularly since the sound source changed course to continue moving towards the animals, even after they exhibited avoidance responses.

Southall et al. (2007a) reviewed the relatively extensive behavioral observations of low frequency cetaceans exposed to non-pulse sources. While there are clearly major areas of uncertainty, Southall et al. (2007a) concluded that the literature indicated that there were no (or very limited) responses to received levels of 90 dB to 120 dB sound pressure level with an increasing probability of avoidance and other behavioral effects in the 120 to 160 dB sound pressure level (received level) range.

While not specific to the low-frequency sonar of the proposed action, both Goldbogen et al. (2013) and Melcon et al. (2012) indicated that behavioral responses to simulated or operational MFA sonar were temporary, with whales resuming normal behavior quickly after the cessation of sound exposure. Further, responses were discernible for whales in certain behavioral states (i.e., deep feeding), but not in others (i.e., surface feeding). As stated in Goldbogen et al. (2013) when summarizing the response of blue whales to simulated MFA sonar, “We emphasize that elicitation of the response is complex, dependent on a suite of contextual (e.g., behavioral state) and sound exposure factors (e.g., maximum received level), and typically involves temporary avoidance responses that appear to abate quickly after sound exposure.” Melcon et al. (2012) reported that blue whales exposed to mid-frequency sonar in the Southern California Bight were less likely to produce low frequency calls (D calls) usually associated with feeding behavior. However,they were unable to determine if suppression of D calls reflected a change in their feeding performance or abandonment of foraging behavior and indicated that implications of the documented responses are unknown. Goldbogen et al. (2013) speculated that if the documented temporary behavioral responses interrupted feeding behavior, this could have impacts on individual fitness and eventually, population health. However, for this to be true, we would have to assume that an individual whale could not compensate for this lost feeding opportunity by either immediately feeding at another location, by feeding shortly after cessation of acoustic exposure, or by feeding at a later time. There is no indication this is the case, particularly since unconsumed prey would likely still be available in the environment in most cases following the cessation of acoustic exposure. Preliminary results from the 2010 to 2011 field season of an ongoing behavioral response study in Southern California waters indicated that in some cases and at low received levels, tagged blue whales responded to mid-frequency sonar but that those responses were mild and there was a quick return to their baseline activity (Southall et al. 2011a). Blue whales responded to a mid-frequency sound source, with a source level between 160 and 210 dB re: 1 µPa at 1 m and a received sound level up to 160 dB re: 1 µPa, by exhibiting generalized avoidance responses and changes to dive behavior during controlled exposure experiments (Goldbogen et al. 2013). However, reactions were not consistent across individuals based on received sound levels alone, and likely were the result of a complex interaction between sound exposure factors such as proximity to sound source and sound type (mid-frequency sonar simulation vs. pseudo-random noise), environmental conditions, and behavioral state. Surface feeding whales did not show a change in behavior during controlled exposure experimentss, but deep feeding and non-feeding whales showed temporary reactions that quickly abated after sound exposure. Whales were sometimes less than a mile from the sound source during controlled exposure experimentss. We would expect similar context dependent, temporary responses from any cetaceans exposed to LFA sonar.

6.2.4 Physiological Stress

Classic stress responses begin when an animal’s central nervous system perceives a potential threat to its homeostasis. That perception triggers stress responses regardless of whether a stimulus actually threatens the animal; the mere perception of a threat is sufficient to trigger a stress response (Moberg 2000; Sapolsky 2006; Seyle 1950). Once an animal’s central nervous system perceives a threat, it mounts a biological response or defense that may consist of a combination of several biological defense responses including, but not limited to, behavioral responses, autonomic nervous system responses, or neuroendocrine responses.

In the case of many stressors, the first and most economical (in terms of biotic costs) response is behavioral avoidance of the potential stressor or avoidance of continued exposure to a stressor. An animal’s second line of defense to stressors involves the autonomic nervous system and the classical “fight or flight” response which includes the cardiovascular system, the gastrointestinal system, the exocrine glands, and the adrenal medulla to produce changes in heart rate, blood pressure, and gastrointestinal activity that humans commonly associate with stress. These responses have a relatively short duration and may or may not have significant long-term effect on an animal’s welfare.

An animal’s third line of defense to a stressor involves its neuroendocrine systems, usually hormones associated with the hypothalmus-pituitary-adrenal system (most commonly known as the HPA axis in mammals or the hypothalamus-pituitary-interrenal axis in fish and some reptiles). Unlike stress responses associated with the autonomic nervous system, virtually all neuroendocrine functions that are affected by stress – including immune competence, reproduction, metabolism, and behavior – are regulated by pituitary hormones. Stress-induced changes in the secretion of pituitary hormones have been implicated in failed reproduction (Moberg 1987; Rivier 1985) and altered metabolism (Elsasser et al. 2000), immune competence (Blecha 2000) and behavior. Increases in the circulation of glucocorticosteroids (cortisol, corticosterone, and aldosterone in marine mammals) are equated with stress.

The primary distinction between stress (which is adaptive and does not normally place an animal at risk) and distress is the biotic cost of the response. During stress an animal uses glycogen stores that can be quickly replenished once the stress is alleviated. In such circumstances, the cost of the stress response does not pose a risk to the animal’s welfare.

However, when an animal has insufficient biotic reserves to satisfy the biotic cost of a stress response, then resources must be shifted away from other biotic functions. When sufficient reserves are diverted from these functions, the functions are impaired. For example, when stress shifts metabolism away from growth, young animals no longer thrive and growth is stunted. When energy is shifted from supporting reproduction, reproductive success is diminished.

In these cases, animals have entered a pre-pathological state and are experiencing “distress” (sensu Seyle 1950) or “allostatic loading” (sensu McEwen and Wingfield 2003). This period of distress will last until the animal replenishes its biotic reserves sufficient to restore normal function.

Relationships between these physiological mechanisms, animal behavior, and the costs of stress responses have also been documented fairly well through controlled experiment; because this physiology exists in every vertebrate that has been studied, it is not surprising that stress responses and their costs have been documented in both laboratory and free-living animals (for examples see, (Holberton et al. 1996; Hood et al. 1998; Jessop et al. 2003; Lankford et al. 2005).

No information has been collected on the physiological responses of marine mammals upon exposure to anthropogenic sounds, studies of other marine animals and terrestrial animals would lead us to expect some marine mammals to experience physiological stress responses and, perhaps, physiological responses that would be classified as “distress” upon exposure to mid-frequency and low-frequency sounds. However, when bowhead and gray whales were exposed to various types of man-made noise, their surfacings become unusually brief with fewer blows per surfacing (Malme et al. 1989; Richardson et al. 1985c; Richardson et al. 1990; Richardson et al. 1986). These authors concluded that changes in surfacing, respiration, and diving behavior may be indicators of stress, although their consequences on the population ecology of the animals that are affected remain unknown.

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. Stress responses to sound exposures have been studied in other animal species, including humans. Jansen (1998) reported on the relationship between acoustic exposures and physiological responses that are indicative of stress responses in humans (for example, elevated respiration and increased heart rates). Jones (1998) reported on reductions in human performance when faced with acute, repetitive exposures to acoustic disturbance. Trimper et al. (1998) reported on the physiological stress responses of osprey to low-level aircraft noise while Krausman et al. (2004) reported on the auditory and physiology stress responses of endangered Sonoran pronghorn to military overflights. Smith et al. (2004a; 2004b) identified noise-induced physiological stress responses in hearing-specialist fish that accompanied short- (TTS) and long-term (PTS) hearing losses. Welch and Welch (1970) reported physiological and behavioral stress responses that accompanied damage to the inner ears of fish and several mammals.

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. 1990a) but showed an increase in catecholamines following exposure to impulsive sounds produced from a seismic water gun (Romano et al. 2004). A bottlenose dolphin exposed to the same seismic water gun signals did not demonstrate a catecholamine response, but did demonstrate an elevation in aldosterone, a hormone that may be a significant indicator of stress in odontocetes (St. Aubin et al. 2001; St. Aubin and Geraci 1989). Increases in heart rate were observed in bottlenose dolphins to which conspecific calls were played, although no increase in heart rate was observed when tank noise was played back (Miksis et al. 2001). Collectively, these results suggest a variable response that depends on the characteristics of the received signal and prior experience with the received signal.

Hearing is one of the primary senses cetaceans use to gather information about their environment and to communicate with conspecifics. Although empirical information on the relationship between sensory impairment (TTS, PTS, and acoustic masking) on cetaceans remains limited, it is reasonable to assume that reducing an animal's ability to gather information about its environment and to communicate with other members of its species would be stressful for animals that use hearing as their primary sensory mechanism. Therefore, we assume that acoustic exposures sufficient to trigger onset PTS or TTS would be accompanied by physiological stress responses because terrestrial animals exhibit those responses under similar conditions (NRC 2003b). Marine mammals might experience stress responses at received levels lower than those necessary to trigger onset TTS (e.g., if exposure caused a significant behavioral response). Based on empirical studies of the time required to recover from stress responses (Moberg 2000), we also assume that stress responses may persist beyond the time interval required for animals to recover from TTS and might result in pathological and pre-pathological states that would be as significant as behavioral responses to TTS.

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.

In summary, 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 and conference report, 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.

6.2.5 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 auditory fatigue, which always results in a localized 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. As a general statement, the more similar the characteristics (i.e., frequency band, duration) of a masking sound are to the sound of interest, the greater its potential for masking.

Acoustic masking from low frequency ocean noise is increasingly being considered as a threat, especially to low-frequency hearing specialists such as baleen whales (Clark et al. 2009). Most underwater low frequency anthropogenic noise is generated by commercial shipping, which has contributed to the increase in oceanic background noise over the past 150 years (Parks et al. 2007a). Shipping noise is primarily in the 20 to 200 Hz frequency band and is increasing yearly (Ross 2005). Andrew et al. (2002) demonstrated an increase in oceanic ambient noise of ten dB sound pressure level since 1963 in the 20 to 80 Hz frequency band as sampled on the continental slope off Point Sur, California, and they ascribed this increase to increased commercial shipping. McDonald et al. (2006a) compared data sets from 1964 to 1966 and 2003 to 2004 for continuous measurements west of San Nicolas Island, California, and found an increase in ambient noise levels of ten to 12 dB sound pressure level in the 30 to 50 Hz band. This increase in low frequency background noise is likely having a widespread impact on marine mammal low-frequency hearing specialists by reducing their access to acoustic information essential for con-specific communication and other biologically important activities, such as navigation and prey/predator detection. Clark et al. (2009) considered this long-term, large-scale increase in low frequency background noise a chronic impact that results in a reduction in communication space, and the loss of acoustic habitat.

Critical ratios 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 1989; Erbe 2000; Johnson 1971).

Clark et al. (2009) developed a methodology for estimating masking effects on communication signals for low frequency cetaceans, including calculating the cumulative impact of multiple noise sources. For example, their technique calculates that in Stellwagen Bank National Marine Sanctuary, when two commercial vessels pass through a North Atlantic right whale's optimal communication space (estimated as a sphere of water with a diameter of 20 km [11 nmi]), 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, but 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 the repertoire of sound production modes used 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. Dahlheim et al. (1984) concluded that gray whales in the San Ignacio Lagoon, Baja, California shifted the frequencies of their vocalizations away from the predominant ambient noise producers in the lagoon to overcome masking effects. In contrast, Biassoni et al. (2001) concluded that the intermittent sounds produced by LFA sonar were unlikely to mask humpback whale songs, but the similarities of its sounds to those produced by the whales could cause some humpback whales to perceive LFA sonar as a competing male. Based on their studies, they concluded that humpback whales could adapt to the presence of LFA sonar and concluded that singing whales could compensate for interference from sound sources like LFA sonar.

Parks et al. (2007a) provided evidence of behavioral changes in the acoustic behaviors of the endangered North Atlantic right whale, and the South Atlantic right whale, and suggested that these were correlated to increased underwater noise levels. The study indicated that right whales might shift the frequency band of their calls to compensate for increased in-band background noise. The significance of their result is the indication of potential species-wide behavioral change in response to gradual, chronic increases in underwater ambient noise. DiLorio and Clark (2010) showed that blue whale calling rates vary in association with seismic sparker survey activity, with whales calling more on days with survey than on days without surveys. They suggested that the whales called more during seismic survey periods as a way to compensate for the elevated noise conditions.

In the presence of LFA sonar, humpback whales have been observed to increase the length of their ‘songs’ (Fristrup et al. 2003; Miller et al. 2000b), 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 shift the frequency content of their calls upward while reducing the rate of calling in areas of increased anthropogenic noise (Parks et al. 2007a) as well as increasing the amplitude (intensity) of their calls (Parks 2009b). 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.

Holt et al. (2009) measured killer whale call source levels and background noise levels in the one to 40 kHz band. The whales increased their call source levels by one dB for every one dB increase in background noise level. A similar rate of increase in vocalization activity was reported for St. Lawrence River belugas in response to passing vessels (Scheifele et al. 2005).

Differential vocal responding in marine mammals has 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), while blue whale feeding/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. Castellote et al. (2012) found that vocalizing fin whales in the Mediterranean left the area where a seismic survey was being conducted and that their displacement persisted beyond the completion of the survey. 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. 2013). 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. 2013). 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. 2007a)), Deruiter et al. (2013) found the responses were variable by species and with the context of the sound exposure.

Evidence suggests that at least some marine mammals have the ability to acoustically identify potential 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. The occurrence of masking or hearing impairment provides a means by which marine mammals may be prevented from responding to the acoustic cues produced by their predators. Whether or not this is a possibility depends on the duration of the masking/hearing impairment and the likelihood of encountering a predator during the time that predator cues are impeded.

Like Richardson et al. (1995d), we assume that baleen whales are very sensitive to low-frequency sounds. As a result, masking effects could occur for many of these whales because they vocalize at low frequencies and are thought to have hearing that is sensitive at the SURTASS LFA sonar frequencies. However, masking effects from SURTASS LFA sonar signals will be limited for a number of reasons. First, the bandwidth of any LFA sonar transmitted signal is limited (30 Hz), and the instantaneous bandwidth at any given time of the signal is small, on the order of approximately ten Hz. Therefore, within the frequency range in which masking is possible, the effect will be limited because animals that use this frequency range typically use signals with greater bandwidths. Thus, only a portion of frequency band for the animal's signal is likely to be masked by the LFA sonar transmissions. Furthermore, when LFA sonar is in operation, the LFA sonar source is active only 7.5 to ten percent of the time (based on historical LFA sonar operational parameters), which means that for 90 to 92.5 percent of the time there is no risk that an animal's signal will be masked by LFA sonar. Therefore, within the area in which energetic masking is possible, any effect of LFA sonar transmissions will be minimal because of the limited bandwidth and intermittent nature of the signal, and the fact that animals that use this frequency region typically produce signals with greater bandwidth that are repeated for many hours.

The echolocation calls of toothed whales are subject to masking by high frequency sound. However, since echolocation calls occur at much higher frequencies than SURTASS LFA sonar, the extent of upward masking (i.e., low frequencies masking high frequencies) would be limited. Additionally, studies on captive odontocetes by Au et al. (1993; 1985; 1974) indicate that some species may use various processes to reduce masking effects (e.g., adjustments in echolocation call intensity or frequency as a function of background noise conditions). There is also evidence that the directional hearing abilities of odontocetes are useful in reducing masking at the high frequencies used for echolocation, but not at the low-moderate frequencies used for communication (Zaitseva et al. 1980).

Although low frequency hearing has not been studied in many odontocete species, those species that have been tested (beluga, killer whale, false killer whale, Risso's dolphin [*Grampus griseus*], and bottlenose dolphin) exhibit poor audiometric and behavioral sensitivity to low frequency sound. For sounds dominated by low frequency components, the maximum radius of audibility for most odontocete species may often be noise-limited when sensitivity is good, and sensitivity-limited when sensitivity is poor. At a maximum 20 percent duty cycle for SURTASS LFA sonar, it is anticipated that any masking of odontocetes would be temporary (i.e., at least 80 percent of the time an animal would be able to perceive incoming signals through low frequency sounds). The possibility of effective masking would only occur for environmental sounds that happen during the ping transmission (maximum 100 seconds) and are at, or at least close to, the frequencies in the 30-Hz-wide bandwidth signal, during the ten seconds the SURTASS LFA sonar was transmitting in that bandwidth. As a result, the available evidence does not lead us to expect masking to directly reduce the reproduction, numbers, or distribution of threatened or endangered odontocetes or elicit behavioral responses that would reduce the reproduction, numbers, or distribution of these species.

Hildebrand (2005a) provided a comparison of anthropogenic underwater sound sources by their annual energy output. On an annual basis, four LFA sonar systems were estimated to have a total energy output of 6.8 x 10^{11} Joules per year. Seismic airgun arrays and mid-frequency military sonars were two orders of magnitude greater, with an estimated annual output of 3.9 and 2.6 x 10^{13} Joules per year, respectively. Super tankers were estimated at 3.7 x 10^{12} Joules per year. Hildebrand (2005a) concluded that anthropogenic sources most likely to contribute to increased underwater noise in order of importance are: commercial shipping, offshore oil and gas exploration and drilling, and naval and other uses of sonar. The percentage of the total anthropogenic acoustic energy budget added by each LFA sonar source is estimated to be 0.21 percent per system (or less), when other man-made sources are considered (Hildebrand 2005a). When combined with the naturally occurring and other man-made sources of noise in the oceans, the intermittent LFA sonar signals barely contribute a measurable portion of the total acoustic energy.

The research reviewed above indicates that masking in marine mammals due to narrowband, intermittent (low duty cycle) LFA sonar signal transmissions is expected to be minimal and unlikely.

6.2.6 Stranding

Stranding is a potential risk for cetaceans. Stranding occurs when marine mammals passively (unintentionally) or purposefully come ashore either alive, but debilitated or disoriented, or dead. Although some species of marine mammals, such as pinnipeds, routinely come ashore during all or part of their life history, stranded marine mammals are differentiated by their helplessness ashore and inability to cope with or survive their stranded situation (i.e., they are outside their natural habitat) (Geraci and Lounsbury 2005).

Strandings of multiple marine mammals, or mass strandings, however, occur only rarely. A mass stranding of marine mammals is the stranding of two or more unrelated cetaceans (i.e., not a mother-calf pair) of the same species coming ashore at the same time and place (Geraci and Lounsbury 2005). Mass strandings typically involve pelagic odontocete marine mammal species that occur infrequently in coastal waters and are usually typified by highly developed social bonds. Marine mammal strandings and mortality events are natural events, and such events continue to occur throughout the world's oceans.

Over the last four decades, marine mammal stranding networks have become established, and the reporting of marine mammal stranding and mortality events has become better documented and publicized. This has led to increased public awareness and concern, especially regarding the potential for anthropogenic causes of stranding and mortality events. Underwater noise, particularly sounds generated by military sonar or geophysical and geologic seismic exploration, has increasingly been implicated as the plausible cause for marine mammal mortality and stranding events. However, despite extensive and lengthy investigations and continuing scientific research, definitive causes or links are rarely determined for the vast majority of marine mammal mass strandings and unusual mortality events (UMEs). It is generally more feasible to exclude causes of strandings or UMEs than to resolve the specific causative factors leading to these events. For instance, although no definitive cause could be identified for the mass stranding and death of 26 common dolphins in the Cornwall region of the United Kingdom during 2008, more than ten factors were excluded or were considered highly unlikely to have caused the stranding (Jepson and Deaville 2009). Ketten (2014) examined the linkage between mass strandings of marine mammals and exposure to military sonar and other high intensity acoustic sources and noted that, in absence of other causative factors, MFA military sonar transmissions have been implicated as a causative factor in several marine mammal strandings in the Bahamas, Hawaii, the Canary Islands, Madeira, and Greece.

As a result of scientific investigations and research over the last decade, especially on beaked whales, the scientific understanding has increased regarding the association between behavioral reactions to natural as well as anthropogenic sources and strandings or deaths of marine mammals. Scientists suspect that for some species, particularly deep-diving marine mammals, behavioral reactions may begin a cascade of physiologic effects, such as gas and fat embolisms, that may result in injury, death, and strandings of marine mammals (Cox et al. 2006; Fernandez et al. 2005a; Zimmer and Tyack 2007).

Globally from 2006 through early 2010, at least 27 mass strandings of 11 marine mammal species occurred. These 27 mass stranding and mortality events were researched and analyzed to substantiate if any occurred within or near SURTASS LFA sonar mission areas, or if any were potentially associated with the transmission of underwater sound from military sonar. Any mass strandings involving beaked whales were also examined, as strandings of this species group have been shown to have a significant correlation with MFA naval sonar activities in some geographic regions (in the Mediterranean and Caribbean Seas but not off the coasts of Japan or Southern California) (Filadelfo et al. 2009). Additionally, marine mammal stranding records from Japan were analyzed for spatial or temporal correlations to LFA sonar operations. The use of SURTASS LFA sonar was not associated with any of the reported 27 mass stranding events or

UMEs that occurred globally between 2006 and early 2010. There is no evidence that LFA sonar transmissions resulted in any difference in the stranding rates of marine mammals in Japanese coastal waters adjacent to LFA sonar mission areas.

For the annual 2014 to 2015 and first two quarters of the 2015 to2016 LOA effective periods, all available media and known stranding databases were monitored for strandings in or adjacent to the western and central North Pacific Ocean mission areas. More details of the marine mammal strandings that occurred during this period can be found in the U.S. Navy's application for renewal of LOAs for 2015 to 2016 (Navy 2015a). During the period from August 2014 to August 2015, 25 marine mammal stranding events occurred in the western and central North Pacific Ocean, respectively, in or adjacent to mission areas for SURTASS LFA sonar in which LFA sonar may have operated during that period. During the first two quarters (August to February) of 2015 to 2016, eight marine mammal strandings have been documented in or adjacent to SURTASS LFA sonar mission areas in the western North Pacific Ocean. Correlating the strandings spatially and temporally with active sonar missions of SURTASS LFA sonar vessels, the U.S. Navy concluded that none of the stranding events reported in the western North Pacific occurred temporally or spatially in conjunction with SURTASS LFA sonar missions. No SURTASS LFA sonar operations occurred in the Hawaiian mission areas during these periods.

As documented above, the employment of LFA sonar is not expected to result in any sonar-induced strandings of marine mammals. Given the large number of natural factors that can result in marine mammal mortality, the high occurrence of marine mammal strandings, and the many years of LFA sonar operations without any reported associated stranding events (greater than12 years), the likelihood of LFA sonar transmissions causing marine mammals to strand is negligible.

6.2.7 Potential 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 vary. 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 exposure on individual marine mammal vitals rates. Further for assessing the impact of acoustic 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 U.S. 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 Population Consequences of Disturbance (PCoD) (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 U.S. 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 (six 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 two-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 three years, and that the inter-calf interval at Abaco in the Bahamas may be four years. New et al. (2013) acknowledge that an assumed two-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 ten percent.

The findings of New et al. (2014) are not applicable to the temporary behavioral disruptions that may be caused by U.S. Navy 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 and conference report to be similarly unaffected. We have no information to suggest animals eliciting a behavioral response (e.g., temporary disruption of feeding) from exposure to SURTASS LFA sonar 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 U.S. Navy activities will not have as long of a duration as those considered in the New et al. (2014) study. No individual animals will be exposed to U.S. Navy activities for a long enough duration to disrupt 50 percent of its annual feeding opportunities over multiple years. New et al. (2014) suggest it would be unlikely even for episodic environmental change, such as El Niño events, to affect the probability of population persistence. As suggested by the authors, the New et al. (2014) model may be more applicable to the consideration of potential long-term behavioral disruptions (e.g., those that may result from climate change).

Until an appropriate quantitative model is developed and until all relevant empirical data is collected to support such a model for the species considered in this opinion and conference report, the best assessment of long-term consequences from U.S. Navy sonar activities will be to monitor marine mammal populations over time.

6.2.8 Interaction of SURTASS LFA Sonar and Other Sound Sources

Several investigators and organizations have expressed concern about the "cumulative impact" (in the NEPA sense of the term) of marine sounds on the ocean environment and its organisms. Any man-made sound that is strong enough to be audible (detectable above natural background noise) will increase total background levels and could interfere with an animal's ability to detect sound signals if the signal is weak, relative to total noise levels. Concern about the cumulative impact of man-made sounds focuses on impacts from individual actions that are insignificant or minor when considered in isolation, but combine to produce effects that are greater than any individual action (either because the effects are synergistic-effects that occur when two or more phenomena interact - multiplicative, or additive). In this opinion, our assessment has focused primarily on the effect of adding LFA sonar to underwater ambient noise levels, but also on the effect of adding mid-frequency active sonar and LFA sonar to underwater ambient noise levels during the same series of exercises.

Richardson et al. (1995d) provided extensive information and arguments about the potential cumulative effects of man-made noise on marine mammals. Those effects included masking, physiological effects and stress, habituation, and sensitization. Those concerns were echoed by Clark and Fristrup (2001), The National Research Council (NRC 2003b), the National Resources Defense Council (Jasny et al. 2005), and others. Although all of these responses have been measured in terrestrial animals reacting to airborne, man-made noises, those studies are counterbalanced by studies of other terrestrial mammals that did not exhibit these responses to similar acoustic stimuli.

Broadband, continuous low-frequency shipping noise is more likely to affect marine mammals than narrowband, low duty cycle SURTASS LFA sonar. Moreover, SURTASS LFA sonar bandwidth is limited (approximately 30 Hz), the average maximum pulse length is 60 seconds, signals do not remain at a single frequency for more than ten seconds, and during an operation the system is off nominally 90 to 92.5 percent of the time. Most mysticetes vocalizations are in the low frequency band below one kHz. No direct auditory measurements have been made for any mysticete, but it is generally believed that their frequency band of best hearing is below 1,000 Hz, where their calls have the greatest energy (Clark 1990; Ketten 2000). However, with the nominal duty cycle of 7.5 to ten percent, masking would be temporary. For these reasons, any masking effects from SURTASS LFA sonar are expected to be negligible and extremely unlikely.

Odontocetes have a broad acoustic range and hearing thresholds measure between 400 Hz and 100 kHz (Finneran et al. 2002b; Richardson et al. 1995d). It is believed that odontocetes communicate above 1,000 Hz and echolocate above 20 kHz (Wursig and Richardson 2009). While the upward spread of masking is known to exist, the phenomenon has a limited range in frequency. Kidd (2007) showed that magnitude of the masking effect decreases as the difference between signal and masking frequency increase; i.e., the masking effect is lower at three times the frequency of the masker than at two times the frequency. Gorga et al. (2002) demonstrated that for a 1.2-kHz masking signal, the upward spread of masking was extinguished at frequencies of six kHz and higher. Therefore, while the phenomenon of upward spread of masking does exist, it is unlikely that LFA sonar would have any significant effect on the hearing of higher frequency animals. Gorga et al. (2002) also demonstrated that the upward spread of masking is a function of the received level of the masking signal. Therefore, a large increase in the masked bandwidth due to upward masking would only occur at high received levels of the LFA sonar signal.

Hildebrand (2005b) provided a comparison of anthropogenic underwater sound sources by their annual energy output. On an annual basis, four LFA sonar systems were estimated to have a total energy output of 6.8 x 10^{11} Joules per year. Seismic airgun arrays and mid-frequency military sonars were two orders of magnitude greater, with an estimated annual output of 3.9 and 2.6 x 10^{13} Joules per year, respectively. Super tankers were estimated at 3.7 x 10^{12} Joules per year. Hildebrand (2005b) concluded that anthropogenic sources most likely to contribute to increased underwater noise in order of importance are: commercial shipping, offshore oil and gas exploration and drilling, and naval and other uses of sonar. The use of LFA sonar is not scheduled to increase beyond the originally analyzed four systems during the next five-year regulation period under the MMPA. The percentage of the total anthropogenic acoustic energy budget added by each LFA sonar source is estimated to be 0.21 percent per system (or less), when other man-made sources are considered (Hildebrand 2005b). When combined with the naturally occurring and other man-made sources of noise in the oceans, the intermittent LFA sonar signals barely contribute a measurable portion of the total acoustic energy.

In a report entitled “Ad-Hoc Group on the Impact of Sonar on Cetaceans,” the International Council for the Exploration of the Sea (ICES 2005) concluded that shipping accounts for more than 75 percent of all human sound in the sea, and sonar amounts to no more than ten percent or so. It further stated that sonar (noise budget) will probably never exceed ten percent, but that sonar deployment seems likely to increase in the future.

In some instances, one or more of the vessels equipped with SURTASS LFA sonar would participate in U.S. Navy training exercises that involve anti-submarine warfare and mid-frequency active sonar. During these exercises, sound fields produced by LFA sonar might co-occur with sound fields generated by mid-frequency active sonar or animals might be exposed to LFA sonar and mid-frequency active sonar in a short time interval. Potential interactions between LFA sonar and other sonars were analyzed on the U.S. Navy’s FSEIS/SOEIS (Navy 2012a). Although the SURTASS LFA sonar and mid-frequency active (MFA) sonars (AN/SQS 53C and AN/SQS 56) are similar in the underlying transmission types, specifically frequency-modulated (FM) sweeps and continuous wave (CW) transmissions, LFA sonar and MFA sonars are dissimilar in other respects. In addition to these multiple differences, the duty cycle, (i.e., the amount of time during sonar operations that the sonar is actually transmitting), is different for SURTASS LFA sonar as opposed to MFA sonar. During SURTASS LFA sonar operations, LFA sonar transmits approximately ten percent of the time (one minute out of ten). During MFA sonar operations, MFA sonar transmits approximately 1.7 percent of the time (1 second out of 60). This means that for any given period of time that both SURTASS LFA sonar and MFA sonars are operating concurrently, the LFA sonar 60-second transmission will be overlapped by one second of MFA transmission, or 1.7 percent of the 60-second LFA sonar ping (one second per 60 seconds). During the ten-minute LFA sonar transmission cycle, the most an animal could be simultaneously exposed from both transmissions is one second for every 600 seconds, or about 0.17 percent of the time that both sonars are operating.

The ocean volumes of Level A harassment received levels for each source are relatively small (1 km [0.54 nmi] radius or less). It is not reasonably foreseeable that SURTASS LFA sonar and MFA sonars would operate simultaneously within ranges less than 9.3 km (5 nmi). Thus, it is not reasonably foreseeable that the Level A harassment volumes of the two sonars could ever overlap during simultaneous transmissions.

The results of two separate analysis methodologies, parametric analysis and underwater acoustic model analysis, were consistent—concurrent MFA/SURTASS LFA sonar operations produce no Level B harassment risk greater than that obtained by simply adding the risks from the individual sources. Therefore, two separate analytic approaches have concluded that there is no potential increase in risk for Level B harassment from concurrent MFA/SURTASS LFA sonar operations. Thus, the potential for this occurring is small and should be considered very conservative. See FSEIS/SOEIS (Navy 2012a) Subchapter 4.7.4 and Appendix E for additional detail.

Because of the potential for seismic surveys to interfere with the reception of passive signals and return echoes, SURTASS LFA sonar operations are not expected to be close enough to these activities to have any synergistic effects. Because of the differences between the LFA sonar coherent signal and seismic air gun impulsive “shots,” there is little chance of producing a “synergistic” sound field. Marine animals would perceive these two sources of underwater sound differently and any addition of received signals would be insignificant. This situation would present itself only rarely, as LFA sonar testing and training operations have not been, and are not expected to be conducted in proximity to any seismic survey activity.

If SURTASS LFA sonar operations were to occur concurrent with other military and commercial sonar systems, synergistic effects are not probable because of differences between these systems. In order for the sound fields to converge, the multiple sources would have to transmit exactly in phase (at the same time), requiring similar signal characteristics, such as time of transmissions, depth, frequency, bandwidth, vertical steering angle, waveform, wavetrain, pulse length, pulse repetition rate, and duty cycle. The potential for this occurring is small.

Therefore, because of major differences in signal characteristics between LFA sonar, MFA, and seismic airguns, there is negligible chance of producing a “synergistic” sound field. It is also unlikely that LFA sonar sources, if operated in proximity to each other would produce a sound field so complex that marine animals would not be able to escape.

6.3 Risk Assessment Framework – Sea Turtles

There are very few studies of the potential effects of underwater sound on sea turtles and most of these examined the effects of sounds of much longer duration or of different types (e.g., seismic airgun) than the SURTASS LFA sonar signals. The analysis of the potential effects on sea turtle species is summarized based on the following SURTASS LFA sonar operational parameters:

- Small number of SURTASS LFA sonar systems to be deployed;
- Geographic restrictions imposed on system employment;
- Narrow bandwidth of the SURTASS LFA sonar active signal (approximately 30 Hz);
- Slowly moving ship, coupled with low system duty cycle, would mean that a sea turtle would spend less time in the LFA sonar mitigation and buffer zones (180-dB sound pressure level sound field); therefore, with a ship speed of less than five knots, the potential for animals being in the sonar transmit beam during the estimated 7.5 to 10 percent of the time the sonar is actually transmitting is very low; and
- Small size of the LFA sonar mitigation zone (180-dB sound pressure level sound field) relative to open ocean areas.

Due to the lack of more definitive data on sea turtle stock distributions in the open ocean, it is not feasible to estimate the percentage of a stock that could be located in a SURTASS LFA sonar operations area at a potentially vulnerable depth, during an LFA sonar sound transmission. Data on sea turtle sound production and hearing are very limited.

6.3.1 Non-Auditory Injury

There are no data on the potential for anthropogenic sound to cause injury in sea turtles. Although not directly related to SURTASS LFA sonar effects, a review of effects of explosives on turtles was done by Viada et al. (2008). For explosive structure removals in the Gulf of Mexico, NMFS specified that the area within 915 m (3,000 ft) of the platform must be clear of sea turtles. Therefore, using a value of 180-dB sound pressure level injury threshold for sea turtles (within approximately 1,000 m [3,281 ft] of the LFA sonar array) is conservative. The probability of a sea turtle being within the 180-dB mitigation zone is considered negligible because of the mitigation measures employed during sonar operations and the five SURTASS LFA sonar operational parameters. Additionally, direct injuries from SURTASS LFA sonar are unlikely because of the relatively lower peak pressures and slower rise times than potentially injurious sources such as explosives. Non-impulsive sources such as sonar also lack the strong shock wave such as that associated with an explosion. In their “Sound Exposure Guidelines for Sea Turtles and Fishes,” a technical report developed and approved by Accredited Standards Committee S3/SC 1 Animal Bioacoustics, Popper et al. (2014) also concluded that the risk of sea turtles experiencing mortality, potential mortal injury, or recoverable injury was low from exposure to low-frequency sonar.

6.3.2 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.

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. A sea turtle would have to be within the LFA sonar mitigation zone (greater than or equal to 180 dB re: 1 µPa [rms] received level) during sonar transmission to be at risk of injury, including permanent loss of hearing (i.e., PTS). Despite the lack of scientific data on the potential effects of low frequency sound on sea turtle hearing and on PTS in sea turtles caused by low frequency sound, the potential for SURTASS LFA sonar to cause PTS in sea turtles can be considered negligible due to the implementation of monitoring and mitigation measures.

Temporary threshold shift is a hearing loss that recovers to the original hearing threshold over a period of time. An animal may not even be aware of a TTS. The animal 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). As with PTS, there are no published scientific data on temporary loss of hearing in sea turtles caused by low frequency sound. In their “Sound Exposure Guidelines for Sea Turtles and Fishes,” a technical report developed and approved by Accredited Standards Committee S3/SC 1 Animal Bioacoustics, Popper et al. (2014) determined that if a sea turtle was located far from the sound source, there was a low risk it could experience TTS, and a moderate risk if the animal was located intermediate or close distances to the source. The five SURTASS LFA sonar operational parameters listed above support the conclusion that the potential for SURTASS LFA sonar to cause TTS in sea turtles is low.

Little is known about how sea turtles use sound in their environment. Based on knowledge of their sensory biology (Bartol and Ketten 2006; Moein Bartol and Musick 2003), sea turtles may be able to detect objects within the water column (e.g., vessels, prey, predators) via some combination of auditory and visual cues. However, research examining the ability of sea turtles to avoid collisions with vessels shows they may rely more on their vision than auditory cues (Hazel et al. 2007). Similarly, while sea turtles may rely on acoustic cues to identify nesting beaches, they appear to rely on other non-acoustic cues for navigation, such as magnetic fields (Lohmann and Lohmann 1996a; Lohmann and Lohmann 1996b) and light (Avens and Lohmann 2003). Additionally, they are not known to produce sounds underwater for communication. As a result, we do not expect instances of threshold shift to have fitness consequences for individual turtles.

6.3.3 Masking

While no masking studies on marine turtles have been conducted, an indirect study looked at the potential for masking by examining sounds in an area known to be inhabited by turtles. These underwater sound recordings were made in one of the major coastal foraging areas for juvenile sea turtles (mostly loggerhead, Kemp's ridley and green sea turtles) in the Peconic Bay Estuary system in Long Island, New York (Samuel et al. 2005). The recording season of the underwater environment coincided with the sea turtle activity season in an inshore area where there is considerable boating and recreational activity, especially during the July to September timeframe. During this time period, received levels at the data collection hydrophone system in the 200 to 700 Hz band ranged from 83 dB re: 1 μPa (rms) (night) up to 113 dB re: 1 μPa (rms) (weekend day). Therefore, during much of the season when sea turtles are actively foraging in New York waters, they are undoubtedly exposed to these levels of noise, most of which is anthropogenic. However, there were no data collected on any behavioral changes in the sea turtles as a consequence of anthropogenic noise or otherwise during this study, so it cannot be stated whether this level of ambient sound would have any physiological and/or behavioral effects on the sea turtles.

Because sound may play a limited role in a sea turtle's environment as described above in section 6.3.2, the potential for masking may be limited. Nonetheless, masking effects may occur for sea turtle species that have critical hearing bandwidths at the same frequencies as the SURTASS LFA sonar. However, masking would likely be temporary. The geographical restrictions imposed on all SURTASS LFA sonar operations would limit the potential for masking of sea turtles in the vicinity of their nesting sites. In summary, masking effects are not expected to be significant because of the nominal 7.5 percent duty cycle[10], the maximum 100-sec signal duration, the fact that the ship is always moving, the limited 30 Hz sonar bandwidth, and the signals not remaining at a single frequency for more than ten seconds.

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[10] Average duty cycle (ratio of sound "on" time to total time) of the SURTASS LFA sonar active transmission mode is less than 20 percent. The typical duty cycle, based on historical LFA sonar operational parameters since 2003 is nominally 7.5 to 10 percent. During the remaining 80 to 92.5 percent of the time, LFA sonar transmitters would be off, thus adding no sound to the water.

6.3.4 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 stressors in addition to 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 Kemp's ridley sea turtles are not found in the 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 from vessel noise and other 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).

6.3.5 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.

Sea turtles can travel many kilometers per day in the open ocean, as shown in tagging studies (Luschi et al. 2003; Papi et al. 1995); and the use of magnetic positional information for long-range navigation has been demonstrated in several diverse animals, including sea turtles (Benhamou et al. 2011; Lohmann et al. 2012). Sea turtles make extensive migrations and movements either for foraging opportunities or to breed. Their migration tracks may extend to thousands of kilometers (Allen 2007; Arendt et al. 2012; Luschi et al. 2006; Shillinger et al. 2008).

Changes in movement patterns or other behaviors due to exposure to a high intensity sound source that causes prolonged displacement of animals from the site of their normal activities could be considered a deleterious effect. Displacement can occur in two dimensions: vertical and horizontal. For example, a sea turtle could move to the surface, where anthropogenic low frequency sound would be weaker, possibly exposing it to a higher degree of predation. As for horizontal displacement, this is probably of greatest importance for non-pelagic sea turtle species (green, olive ridley, hawksbill), for which displacement from preferred benthic habitats could be construed as more serious.

Behavioral responses to human activity have been investigated for only a few species of sea turtles: green and loggerhead (McCauley et al. 2000; O'Hara and Wilcox 1990); and olive ridley, leatherbacks, loggerhead, and 160 unidentified turtles (hard-shell species) (Weir 2007). The work by O’Hara and Wilcox (1990) and McCauley et al. (2000) reported behavioral changes of sea turtles in response to seismic airguns. O’Hara and Wilcox (1990) reported avoidance behaviors by loggerheads in response to airguns with sound levels (received level) of 175 to 176 dB re: 1 µPa (peak-to-peak). McCauley et al. (2000) reported noticeable increases in swimming behavior for both green and loggerhead turtles at received levels of 166 dB re: 1 µPa (peak-to-peak). At 175 dB re: 1 µPa (peak-to-peak) received level, both green and loggerhead sea turtles displayed increasingly erratic behavior (McCauley et al. 2000). In their “Sound Exposure Guidelines for Sea Turtles and Fishes,” a technical report developed and approved by Accredited Standards Committee S3/SC 1 Animal Bioacoustics, Popper et al. (2014) determined that the relative risk of sea turtles responding behaviorally to low-frequency sonar is low, regardless of the distance from the source.

Also, 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. 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, 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 and conference report.

Based on the hearing data, if a sea turtle happened to be in proximity of a SURTASS LFA sonar operations area, it could hear the low frequency transmissions. Given that the majority of sea turtles encountered would probably be transiting in the open ocean from one site to another, the possibility of significant displacement would be unlikely.

6.4 Exposure Analysis

Here we estimate the potential exposure of ESA-listed species or designated critical habitat to the stressors we determined posed a risk to ESA-listed species. Because of the reasons provided in Section 6.1, this exposure analysis only addresses exposure to signals generated by SURTASS LFA sonar. Our analyses consider the contributions of the intensity of received levels, the duration of a potential exposure, and how frequently individuals of an ESA-listed species might be exposed at different received levels. We assumed that exposure to SURTASS LFA sonar, regardless of the sound pressure level, is a pre-requisite for ESA-listed species or designated critical habitat to be adversely affected by the sonar.[4]

This section describes the conditions under which ESA-listed species could be exposed to LFA sonar based on evaluations of the available information and also describes potential relationships between differing levels of exposure to LFA sonar and potential effects on ESA-listed species. Therefore, we assume that the potential biological risk associated with exposure to LFA sonar is a function of an animal’s exposure to a sound and their response to that exposure, that could adversely affect the animal’s hearing, behavior, psychology, or physiology.

As discussed previously, the LFA sonar system consists of up to 18 low-frequency acoustic-transmitting projectors that are suspended from a cable beneath a ship. The source level of an individual projector in the LFA sonar array is approximately 215 dB, and the sound field of the array can never have a sound pressure level higher than that of an individual projector. The typical LFA sonar signal is not a constant tone, but is a transmission of various signal types that vary in frequency and duration (including continuous wave and frequency-modulated signals). The U.S. Navy refers to a complete sequence of sound transmissions as a “ping” which can range from between six and 100 seconds, with no more than ten seconds at any single frequency. The time between pings will typically range from six to 15 minutes. The U.S. Navy has stated that the duty cycle (the ratio of sound “on” time to total time) would not be greater than 20 percent. Based on operations since 2003, the average duty cycle for the LFA sonar system has been 7.5 to ten percent (Navy 2012a).

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[4] We recognize that LFA sonar could have indirect, adverse effects on species or critical habitat by disrupting marine food chains, a species predator, or species competitors; however, we could not identify a situation where this concern would be applicable to species under NMFS’ jurisdiction.

6.4.1 Intensity

Sound transmissions are usually measured in terms of sound pressure levels, which are denoted as decibels and which have a reference pressure value of 1μPa. The logarithmic nature of the dB scale means that each ten dB increase is a ten-fold increase in power (e.g., 20 dB is a 100-fold increase, 30 dB is a 1,000-fold increase). Humans perceive a ten dB increase in noise as a doubling of sound level, or a ten dB decrease in noise as a halving of sound level.

As described in Section 6.1.2.1, implementing the LFA sonar mitigation and buffer zone which are approximately two km (1.2 mi; 1.1 nmi) around the vessel will ensure that no marine mammals are exposed to a sound pressure level greater than approximately 175 dB re: 1μPa. For an animal to be exposed at received levels greater than 180 dB re: 1µPa (rms), the animal would have to occur in the same approximately one km radius 180 dB re: 1µPa (rms) sound field of the LFA sonar transmit array (centered at approximate depth of 122 m [400 ft]), would have to enter without being detected, and would have to remain there when the LFA sonar transmitter was operating. Based on the available information, we believe the probability of all of these events occurring, although possible, is extremely improbable.

Further, SURTASS LFA sonar will be operated to ensure that sonar sound fields do not exceed 180 dB re: 1 μPa (rms) within 22 km (12 nmi) of any coastline, including offshore islands, or designated offshore areas that are biologically important for marine mammals outside the 22 km zone during seasons specified for a particular area. When in the vicinity of known recreational and commercial dive sites, SURTASS LFA sonar would be operated to ensure that the sound field at these sites would not exceed 145 dB re: 1 µPa (rms), adding an additional level of protection for marine mammals, sea turtles, and fish located within designated dive sites.

Thus far, the combination of geographic operating constraints, operating protocols, monitoring/mitigation measures, and shut-down procedures appear to have prevented most all threatened and endangered species of marine mammal and sea turtles from being exposed to SURTASS LFA sonar at received levels exceeding 180 dB. Further, these factors have prevented listed species from being exposed in certain areas that are critical to their ecology, critical to large portions of their populations, or both. The U.S. Navy proposes to continue using these measures in the prescribed manner. Therefore, based on the evidence available, we conclude that it would be very unlikely for an ESA-listed marine animal to be exposed to received levels of LFA sonar at or above 180 dB re: 1 µPa (rms).

6.4.2 Exposure Duration

The U.S. Navy proposes to conduct active operations of SURTASS LFA sonar systems on up to four vessels for 240 days. The maximum time of active sonar transmissions per vessel would be 432 hours each year (based on a 7.5 percent duty cycle). The duration of a typical SURTASS LFA sonar ping would range from six to 100 seconds, with no more than ten seconds at a single frequency; intervals between pings would range from six to 15 minutes. Pings would consist of various signal types that vary in frequency (between 100 and 500 Hz) and duration (including continuous wave and frequency-modulated signals). When the system is turned off, no additional energy would enter the ocean's environment.

The duration of an animal's exposure to SURTASS LFA sonar signals will depend on their proximity to the transmitter and their location in the water column. Nevertheless, although highly improbable, because of the length of individual pings, individual animals could be exposed to SURTASS LFA sonar transmissions for periods ranging from six to 100 seconds at a time.

6.4.3 Exposure Frequency

Individual animals or groups of animals have a low statistical probability of being exposed to SURTASS LFA sonar signals on multiple, separate occasions. However, the number of times an animal could be exposed to sound pressure levels associated with SURTASS LFA sonar transmissions will depend on the deployment schedule of the SURTASS LFA sonar vessels.

Some individuals may be exposed to multiple anthropogenic 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 and conference report, we do not believe this will result from the operation of SURTASS LFA sonar. 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. 2011). 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 556 km (300 nmi) 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 U.S. Navy MFA 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 U.S. Navy's Southern California 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 U.S. Navy has been intensively training with MFA 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 U.S. Navy began using MFA sonar more than 60 years ago, there are no data available to suggest that beaked whale numbers have declined on the range where U.S. Navy sonar use has routinely occurred. As Moore and Barlow (2013) point out, it remains clear that the U.S. 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 U.S. Navy does not conduct in-water training with MFA sonar.

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). Assessing the effects of sounds, both individually and cumulatively, on marine species is difficult because responses depend on a variety of factors including age class, prior experience, behavioral state at the time of exposure, and indirect effects. Responses may be also be influenced by other non-sound related factors (Ellison et al. 2012a; Goldbogen et al. 2013; Kight and Swaddle 2011; McGregor 2013; Read et al. 2014a; Williams et al. 2014a) . McGregor (2013) summarized sound impacts and described two types of possible effects based on the studies they reviewed: 1) an apparent effect of noise on communication, but with a link between demonstrated proximate cost and ultimate cost in survival or reproductive success being inferred rather than demonstrated, and 2) studies showing a decrease in population density or diversity in relation to noise, but with a relationship that is usually a correlation, so factors other than noise or its effect on communication might account for the relationship. Within the ocean environment, aggregate anthropogenic impacts have to be considered in context of natural variation and climate change (Boyd and Hutchins 2012). These contexts can include additive effects from two or more factors, multiplicity where response from two or more factors is greater than the sum of individual effects, synergism between factors and response, antagonism as a negative feedback between factors, acclimation as a short-term individual response, and adaptation as a long-term population change (Boyd and Hutchins 2012). To address aggregate impacts and responses from any changes due to processes such as habituation, tolerance, and sensitization, future experiments over an extended period of time still need further research (Bejder et al. 2009; Blickley et al. 2012) (Read et al. 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, it would be possible for individuals confined to a specific area to be exposed to acoustic stressors (e.g., LFA 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 120 dB). For example, Goldbogen et al. (2013) indicated some horizontal displacement of deep foraging blue whales in response to simulated MFA sonar. Given these animal's mobility and large ranges, we would expect these individuals to temporarily select alternative foraging sites nearby until the exposure levels in their initially selected foraging area have decreased. Therefore, even temporary displacement from initially selected foraging habitat is not expected to impact the fitness of any individual animals because we would expect equivalent foraging to be available in close proximity. Because we do not expect any fitness consequences from any individual animals, we do not expect any population level effects from these behavioral responses.

6.4.4 Mitigation Measures to Minimize the Likelihood of Exposure

The U.S. Navy proposes several measures to reduce or eliminate the likelihood of exposing marine mammals to LFA sonar transmissions at high decibel levels including a monitoring program (visual, passive sonar and the HF/M3 active sonar system), delay/suspension protocols for LFA sonar transmissions, and geographic constraints on the use of SURTASS LFA sonar (coastal mitigation, LFA sonar mitigation and buffer zones, OBIAs, and avoidance of known recreational and commercial dive sites).

6.4.4.1 Visual Monitoring

The effectiveness of visual monitoring is limited to daylight hours, and its effectiveness declines during poor weather conditions. In line transect surveys, the range of visual sighting effectiveness (distance from the ship's track, called effective strip width) varies with an animal's size, group size, reliability of conspicuous behaviors (blows), pattern of surfacing behavior, and positions of the observers (which includes the observer's height above the water surface). For most large baleen whales, effective strip width can be about 3 km (1.6 nmi) up through Beaufort sea state six (Buckland and Borchers 1993). For harbor porpoises the effective strip width is about 250 m (273 yd), because they are much smaller and less demonstrative on the surface than the baleen whales (Palka 1996). The percentage of animals that will pass unseen is difficult to determine, but for minke whales, Schweder et al. (1992) estimated that visual survey crews did not detect about half of the animals in a strip width. Palka (1996) and Barlow (1988) estimated that visual survey teams did not detect about 25 percent of the harbor porpoises in a strip width. Visual detections of marine mammals during SURTASS LFA sonar mission from 2002 to February 14, 2015 totaled 12 sightings.

6.4.4.2 Passive Acoustic Monitoring

The effectiveness of passive acoustic detection is considered to be higher than visual monitoring. Thomas et al. (1986) and Clark and Fristrup (1997) concluded that the effective strip width and detection rates for passive acoustic monitoring is greater than that for visual, but the percentage of animals that will be undetected by the methods is unknown. Frequency coverage for this mitigation method using the SURTASS passive array is between 0 and 500 Hz, so vocalizing animals are more likely to be detected than animals that do not vocalize. This would increase the detection rate of gray, humpback, fin, blue, and minke whales, and some of the beaked whale and dolphin species. Passive acoustic detections of marine mammals during SURTASS LFA sonar missions from 2002 to February 14, 2015, totalled eight animals.

6.4.4.3 High Frequency Marine Mammal Monitoring Sonar

The HF/M3 active sonar system appears to be the most effective measure the U.S. Navy uses to detect animals within two kms of the projectors. Recent testing of the HF/M3 active sonar system demonstrated a probability of single-ping detection above 95 percent within the LFA sonar mitigation zone for many marine mammals (Ellison and Stein 1999). Under normal operating conditions, marine mammals will receive multiple pings, increasing the probability of detection to near 100 percent (Navy 2012a). If any of these monitoring methods detects animals within the LFA sonar mitigation or buffer zones, the projectors would be shut down until the animal(s) move out of the mitigation and buffer zones. Combined with the visual monitoring and passive acoustic monitoring protocols, this should minimize the risk of marine mammals being exposed to sound pressure levels at or above 180 dB re: 1 µPa (rms) at 1 m.

The HF/M3 monitoring detected 151 marine mammals during SURTASS LFA sonar missions from 2002 to February 14, 2015, and these detections resulted in the delay or suspension of SURTASS LFA sonar transmissions. The U.S. Navy followed the appropriate protocols and LFA sonar transmissions were suspended or delayed due to mitigation protocols. As a result, marine mammals that had been detected were not exposed to sonar transmissions at received levels above 180 dB.

The HF/M3 system also increases the likelihood of detecting marine turtles and large fish, such as sturgeon and sharks. Because the HF/M3 sonar is positioned at the top of the LFA vertical array, sea turtles, sturgeon, and scalloped hammerhead sharks would first have to swim through the one-km HF/M3 sonar detection zone (*i.e.*, the very same one-km buffer zone for marine mammals) where detection is highly likely, before entering the LFA sonar mitigation zone.

HF/M3 sonar ramp-up also decreases the likelihood an ESA-listed marine species would be exposed to high sound pressure levels. Benda-Beckmann et al. (2014) investigated the effectiveness of ramp-up procedures in reducing the area within which changes in hearing thresholds can occur. They modeled the level of sound killer whales (*Orcinus orca*) exposed to a generic sonar operation preceded by different ramp-up schemes. Results indicate that ramp-up procedures reduced the risk of killer whales receiving sounds of sufficient intensity to affect their hearing. The main factors that limited effectiveness of ramp-up in a typical antisubmarine warfare scenario were high source level, rapid moving sonar source, and long silences between consecutive sonar transmissions.

6.4.4.4 LFA Sonar Mitigation and Buffer Zones

Inside the LFA sonar mitigation and buffer zones during a ping, ESA-listed species could be exposed to sound levels at or above 180 dB and could experience threshold shifts (loss in hearing sensitivity) or more serious auditory injury. However, the LFA sonar mitigation and buffer zones were established and designed to prevent marine mammal or sea turtles from being exposed to energy levels high enough to produce these outcomes. Several aspects of the LFA sonar mitigation and buffer zones reduce an ESA-listed species' chances of being exposed to LFA sonar at sound pressure levels at or above 180 dB; (a) the mitigation zone extends approximately 0.75 to 1.00 km (0.40 to 0.56 nmi) from the transmitters and is supplemented by a one-km buffer zone, (b) the detection probabilities associated with the HF/M3 sonar (based on a single ping) exceed 95 percent for small dolphins at about 750 m (2,460 ft), whale calves at 1,000 m (3,281 ft) and large whales at more than 1,500 m (44,921 ft); (c) during actual operations marine mammal or larger sea turtles will receive approximately five pings from the HF/M3 sonar with the probability of detection t approaching 100 percent; (d) the depth of the LFA sonar transmitters would reduce the risk of exposing animals located near the ocean's surface or in the mixed layer; and (e) marine mammals and larger sea turtles have a high probability of being detected within the LFA sonar mitigation and buffer zones and, as a result, a low probability of being exposed to sound pressure levels greater than 180 dB re: 1 µPa (rms) at 1 m.

For an animal to be exposed to LFA sonar transmissions at sound pressure levels greater than 180 dB, the animal would have to enter the LFA sonar mitigation and buffer zones without being detected by marine mammal observers or the HF/M3 sonar system and would have to remain in the LFA sonar mitigation and buffer zones when LFA sonar transmitters were operating. The monitoring results of the deployments of the SURTASS LFA sonar system from 2002 through May of 2013 support this conclusion. The active acoustic monitoring (HF/M3 sonar) resulted in a total of 153 possible marine mammal/sea turtle contacts that resulted in suspensions of operations in accordance with the terms and conditions of the existing permit. No contacts were reported within the 180-dB LFA sonar mitigation zone during transmissions, which suggests that no marine mammals were subjected to LFA sonar exposures that might have resulted in injuries. It’s possible that a very small percentage of animals when undetected, but monitoring information does not suggest this is likely.

6.4.4.5 Geographic Restrictions

The SURTASS LFA sonar system would be operated in a manner that would not cause sonar sound fields to exceed 180 dB re: 1 µPa (rms) within the 22 km (12 nmi) coastal standoff range. In addition, the LFA sonar fields will not exceed 180 dB re: 1 µPa (rms) in an OBIA or within the 1-km buffer zone seaward of any OBIA boundary during biologically important seasons. When in the vicinity of known recreational and commercial dive sites, SURTASS LFA sonar will be operated to ensure that the sound field at these sites would not exceed 145 dB re: 1 µPa (rms).

These operational constraints would further reduce the likelihood of exposing threatened and endangered species of whales to SURTASS LFA sonar at sound pressure levels exceeding 180 dB re: 1 µPa (rms) in some areas that are critical to their ecology, critical to large portions of their populations, or both.

6.4.5 Exposure Estimates

For the recent FSEIS/SOEIS (Navy 2012a) and third MMPA rulemaking, the U.S. Navy modeled 19 worldwide ocean areas where SURTASS LFA sonar operations could occur. The U.S. Navy used AIM, as further described in Section 3.8.1 and the 2012 SURTASS LFA Sonar FSEIS/SOEIS. Since the U.S. Navy could not model every part of the ocean waters in which SURTASS LFA sonar operations could potentially occur, 19 sites relevant to U.S. national security interests were selected as nominal examples. The U.S. Navy proposes to conduct 20 missions using SURTASS LFA sonar in the following areas (Table 27.):

- Up to 16 missions in the western North Pacific Ocean, which includes the following operational areas: east of Japan; North Philippine Sea; west Philippine Sea; offshore Guam; Sea of Japan; East China Sea; South China Sea; Japan (25 to 40° North and 10 to 25° North), and northeast of Japan.

- Up to two missions in the central North Pacific Ocean, which includes the northern and southern Hawaii mission areas.

- Up to two missions in the Indian Ocean, which includes the Arabian Sea, Andaman Sea, and northwest of Australia.

Under the 2012 MMPA Final Rule annual LOAs, all missions are conducted to ensure that no more than 12 percent of any marine mammal stock, including those of ESA-listed species, would be taken by Level B harassment annually over the course of the five-year regulation, regardless of the number of LFA sonar vessels operating.

Table 27. Number of proposed SURTASS LFA sonar missions by area proposed to occur 15 August 15, 2016 through 14 August 14, 2017.

Mission Area Number	Mission Area - Modeled Site	Number of Missions
1	East of Japan	1.5
2	North Philippine Sea	3
3	West Philippine Sea	3
4	Offshore Guam (MIRC)	3
5	Sea of Japan	11
6	East China Sea	1
7	South China Sea	1
8	Offshore - Northwest Pacific 25 to 40° North (Offshore Japan)	1
9	Offshore Northwest Pacific 10 to 25° North (Offshore Japan)	1
10	Hawaii North	1
11	Hawaii South	1
12	Arabian Sea	0.5
13	Andaman Sea	1
14	Northwest of Australia	0.5
15	Northeast of Japan	0.5

Modeling to determine impacts from exposure to LFA sonar transmissions on marine mammals was based on the number of missions per mission area (Table 26) for a nominal seven-day mission. Nominal operational assumptions for the model were a mission lasting seven days with an LFA sonar duty cycle of 7.5 percent, which is equal to 12.6 hours of LFA sonar transmission per mission (Table 28).

The U.S. Navy defined the basement value for the risk model, also called the “B parameter”, below which the risk is so low that calculations are impractical as 119-dB. This level is taken as the estimate of received level below which the risk of significant change in a biologically important behavior approaches zero for the SURTASS LFA sonar risk assessment. This level is the value at which avoidance reactions have been noted in bowhead, beluga, and gray whales (which are mitigated by geographic restrictions on SURTASS LFA sonar operations). We recognize that for actual risk of changes in behavior to be zero, the signal-to-noise ratio at the animal must also be zero. However, the present convention of ending the risk calculation at 119 dB single ping equivalent has a negligible impact on subsequent calculations, because the risk function does not attain appreciable values until received levels exceed 130 dB single ping equivalent. Hence, the values estimated in the analysis (Table 29) are the proportion of each species stock that would be expected to be exposed to SURTASS LFA sonar levels between 119 and 180 dB.

Table 28. Critical Parameters Used in the U.S. Navy SURTASS LFA Sonar Model.

LFA sonar Acoustic Source Parameters	Value
Vertical line array elements	18
Nominal source level for single element	215 dB re: 1 µPa at 1 m (rms) (SPL)
Nominal effective source level for array	227.5 dB re: 1 µPa at 1 m (rms) (SPL)
Nominal frequency	300 Hz
Signal bandwidth	Approximately 30 Hz
Nominal duty cycle	7.5 percent
Array depth	87 to 157 m (285 to 515 ft)
Array center depth	122 m (400 ft)
Acoustic Integration Model (AIM) Input	
Species/stock name/stock abundance/density	animals per km²
Coordinates of operating area	Latitude/Longitude
Season	Summer/fall/winter/spring
Mission length	7 days (168 hours)
Active transmission time (based on 7.5 percent duty cycle)	12.6 hours per 7-day mission
Vessel speed	7.4 km per hour (4 knots)
Ship movement	triangular path, 8 hours per leg
Animat/Species Specific AIM Input	
Surface time (min/max)	Heading variable (angle/time)
Surface/dive angle	Swim speed (min/max)
Dive depth (min/max)	Swim speed distribution (normal, gamma)
Dive duration (min/max)	
LFA sonar Acoustic Propagation Modeling	
Standard Parabolic Equation Model for transmission loss versus depth and range from source	
Gridded Global Relief Data from the NOAA National Geophysical Data Center (NGDC) for bathymetry	
Generalized Digital Environmental Model (GDEM) (OAML 2000) for sound velocity profiles for each	

location and season
Bechmann-Spezzichino formula modified by Leibiger (1978) to calculate surface losses using a wind speed of 15 knots
Consolidated bottom loss upgrade (CBLUG) database (OAML 2000) for bottom loss

Table 29. Percentage estimated of each ESA-listed marine mammal stock that may be affected by the operation of SURTASS LFA sonar at RLs of 120 to 180 dB SEL, based on modeling of seven-day missions in each potential mission area (number of missions by mission area listed in table).

Species by Mission Area	Marine Mammal Stock or DPS	Estimated Number in Stock	Total Percent Stock Affected <180 dB SPE for All Missions
East of Japan – 1.5 mission			
Blue whale	WNP	9,250	<0.01
Fin whale	WNP	9,250	0.02
Humpback whale	WNP	1,328	0.42
North Pacific right whale	WNP	922	0.01
Sei whale	NP	7,000	0.22
Sperm whale	NP	102,112	0.02
North Philippine Sea – 3 missions			
Blue whale	WNP	9,250	0.01
Fin whale	WNP	9,250	0.10
Humpback whale	WNP	1,328	3.27
North Pacific right whale	WNP	922	0.05
Sperm whale	NP	102,112	0.06
West Philippine Sea – 3 missions			
Blue whale	WNP	9,250	0.01
Fin whale	WNP	9,250	0.11
Humpback whale	WNP	1,328	4.94
Sperm whale	NP	102,112	0.03
Offshore Guam – 3 missions			
Blue whale	WNP	9,250	<0.01
Fin whale	WNP	9,250	<0.01
Humpback whale	WNP	1,328	0.72
Sei whale	NP	7,000	0.07
Sperm whale	NP	102,112	0.03
Sea of Japan – 1 missions			
Fin whale	WNP	9,250	0.33
North Pacific right whale	WNP	922	0[1]
Western North Pacific gray	WNP	140	0.03
Sperm whale	NP	102,112	0.03
Spotted Seal	Southern	3,500	<0.01
East China Sea – 1 mission			

Species by Mission Area	Marine Mammal Stock or DPS	Estimated Number in Stock	Total Percent Stock Affected <180 dB SPE for All Missions
Fin whale	ECS	500	0.75
North Pacific right whale	WNP	922	0.02
Western North Pacific gray	WNP	140	0.11
Sperm whale	NP	102,112	0.01
Spotted seal	Southern	1,000	0.02
South China Sea – 1 mission			
Fin whale	WNP	9,250	0.03
Humpback whale	WNP	1,328	0.43
North Pacific right whale	WNP	922	0.03
Western North Pacific gray	WNP	140	0.03
Sperm whale	NP	102,112	0.01
Offshore North Pacific (25 to 40° North) (Offshore Japan) – 1 missions			
Blue whale	WNP	9,250	<0.01
Fin whale	WNP	9,250	<0.01
Humpback whale	WNP	1,328	0.11
Sei whale	NP	7,000	0.09
Sperm whale	NP	102,112	0.03
Hawaiian monk seal	Hawaiian	1,153	0.26
Offshore North Pacific (10 to 25° North) (Offshore Japan) – 1 missions			
Blue whale	CNP	9,250	<0.01
Fin whale	WNP	9,250	<0.01
Humpback whale	WNP	1,328	0.64
Sei whale	NP	7,000	0.98
Sperm whale	NP	102,112	0.02
Hawaii North – 1 mission			
Blue whale	CNP – Hawaii	81	0.87
Fin whale	Hawaii	58	0.75
Sei whale	Hawaii	178	1.29
Humpback whale	CNP	10,103	0.12
Sperm whale	Hawaii	3,354	0.50
False Killer Whale	MHI Insular DPS	151	1.27
Hawaiian monk seal	Hawaiian	1,153	0.11
Hawaii South – 1 mission			
Blue whale	CNP – Hawaii	81	0.66
Fin whale	Hawaii	58	0.56
Sei whale	Hawaii	178	0.91
Humpback whale	CNP	10,103	0.14
Sperm whale	Hawaii	3,354	0.42

Species by Mission Area	Marine Mammal Stock or DPS	Estimated Number in Stock	Total Percent Stock Affected <180 dB SPE for All Missions
False Killer Whale	MHI Insular	151	0.05
Hawaiian monk seal	Hawaiian	1,153	0.12
Arabian Sea – 0.5 mission			
Blue whale	North Indian	3,432	<0.01
Fin whale	Indian	1,716	0.07
Humpback whale	Arabian Sea	200	0.01
Sperm whale	North Indian	24,446	0.05
Andaman Sea – 1 missions			
Blue whale	North Indian	3,432	0.01
Fin whale	Indian	1,716	<0.01
Sperm whale	North Indian	24,446	0.05
Northwest of Australia – 0.5 missions			
Blue whale	South Indian	1,657	0.03
Fin whale	South Indian	38,185	0.03
Humpback whale	WAU	14,000	<0.01
Sperm whale	South Indian	24,446	0.02
Northeast of Japan – 0.5 missions			
Blue whale	WNP	9,250	0[1]
Fin whale	WNP	9,250	0.03
Humpback whale	WNP	1,328	0.62
North Pacific right whale	WNP	922	0.11
Sei whale	NP Pelagic	7,000	0.45
Western North Pacific gray	WNP	140	<0.01
Sperm whale	NP	102,112	0.07
Steller sea lion	Western	82,516	<0.01

[1] Species is not expected in the mission area during that season.

WNP = Western North Pacific
CNP = Central North Pacific
NP = North Pacific
ECS = East China Sea
WAU = West Australia

6.4.5.1 Blue Whale

Blue whales may be exposed to and affected by SURTASS LFA sonar during a proposed 17 missions in 11 mission areas from August 2016 to August 2017 (Table 30).

Table 30. Estimated blue whale exposures to SURTASS LFA sonar at RLs between 120 and 180 dB SPL during the 2016 to 2017 annual period.

Site Number	Operating Area	Number of Missions	Marine Mammal Stock	Number in Stock	Percent Stock Affected to 120 to 180 dB[1]	Estimated Number of Instances of Behavioral Harassment [2]	Estimated Exposures without Criteria Processing[3]
1	East of Japan	1.5	WNP	9,250	<0.01	2	34
2	North Philippine Sea	3	WNP	9,250	<0.01	2	61
3	West Philippine Sea	3	WNP	9,250	<0.01	3	99
4	Offshore Guam	3	WNP	9,250	<0.01	2	`6
8	Offshore North Pacific (25 to 40° North) (Offshore Japan)	1	WNP	9,250	<0.01	1	34
9	Offshore North Pacific (10 to 25° North) (Offshore Japan)	1	WNP	9,250	<0.01	1	32
15	Northeast of Japan	0.5	WNP	9,250	N/A[4]	N/A	0
Total Western North Pacific Stock					0.02	11	276
10	Hawaii North (25° North, 158° West)	1	CNP - Hawaii	81	0.87	1	110
11	Hawaii South (19.5° North 158.5° West)	1	CNP - Hawaii	81	0.66	1	19
Total Central North Pacific-Hawaii Stock					1.53	2	129
12	Arabian Sea	0.5	North Indian	3,442	<0.01	1	19
13	Andaman Sea	1	North Indian	3,442	0.01	1	14
Total North Indian Stock					0.01	2	33
14	Northwest of Australia	0.5	South Indian	1,657	0.03	1	19

Total South Indian Stock					0.03	1	19
	Totals	**17**			**1.59**	**16**	**457**

[2] This column represents the estimated number of animals that will exhibit a behavioral response (i.e., be affected) by exposure to SURTASS LFA sonar between 120 and 180 dB SPL during the annual period. The SURTASS LFA sonar risk continuum, as described in Section 3.8.1.4, was used to determine how many exposures between 120 and 180 dB SPL would be expected to result in a behavioral response.

[3] This column represents the estimated number of exposures that will occur between 120 and 180 dB SPL (rms) during the 2016 to 2017 annual period. Fractional animals wpotentially affected have been rounded up to the next whole number.

[4] Not expected in the mission area during that season.

6.4.5.2 Fin Whale

Fin whales may be exposed to and affected by SURTASS LFA sonar during a proposed 20 missions in 15 mission areas from August 2016 to August 2017 (Table 31).

Table 31. Estimated fin whale exposures to SURTASS LFA sonar at RLs between 120 and 180 dB SPL during the 2016 to 2017 annual period.

Site Number	Operating Area	Number of Missions	Marine Mammal Stock	Number in Stock	Percent Stock Affected 120 to 180 dB[1]	Estimated Number of Instances of Behavioral Harassment[2]	Estimated Exposures without Criteria Processing[2]
1	East of Japan	1.5	WNP	9,250	0.02	3	426
2	North Philippine Sea	3	WNP	9,250	0.10	11	1,202
3	West Philippine Sea	3	WNP	9,250	0.11	11	1,294
4	Offshore Guam	3	WNP	9,250	<0.01	2	16
5	Sea of Japan	1	WNP	9,250	0.33	31	2,527
7	South China Sea	1	WNP	9,250	0.03	4	511
8	Offshore North Pacific (25 to 40° North) (Offshore Japan)	1	WNP	9,250	<0.01	1	341
9	Offshore North	1	WNP	9,250	<0.01	1	32

	Pacific (10 to 25°North) (Offshore Japan)						
15	Northeast of Japan	0.5	WNP	9,250	0.22	21	669
Total Western North Pacific Stock					0.81	85	7,018
6	East China Sea	1	ECS	500	0.75	4	183
Total East China Sea Stock					0.75	4	183
10	Hawaii North (25° North, 158° West)	1	Hawaii	58	0.75	1	73
11	Hawaii South (19.5° North 158.5° West)	1	Hawaii	58	0.56	1	13
Total Hawaii Stock					1.31	2	86
12	Arabian Sea	0.5	North Indian	1,716	0.07	2	417
13	Andaman Sea	1	North Indian	1,716	<0.01	1	5
Total Northern Indian Stock					0.07	3	422
14	Northwest of Australia	0.5	South Indian	38,185	0.03	13	783
Total Southern Indian Stock					0.03	13	783
	Totals	**20**			**0.03**	**107**	**8,492**

[1] Percent stock has been rounded up to two decimal points.

[2] This column represents the estimated number of animals that will exhibit a behavioral response (i.e., be affected) by exposure to SURTASS LFA sonar between 120 and 180 dB SPL during the annual period. The SURTASS LFA sonar risk continuum, as described in Section 3.8.1.4, was used to determine how many exposures between 120 and 180 dB SPL would be expected to result in a behavioral response.

[3] This column represents the estimated number of exposures that will occur between 120 and 180 dB SPL (rms) during the 2016 to 2017 annual period. Fractional animals wpotentially affected have been rounded up to the next whole number.

6.4.5.3 Western North Pacific Gray Whale

Western North Pacific gray whales may be exposed to and affected by SURTASS LFA sonar during a proposed 3.5 missions in four mission areas from August 2016 to August 2017 (Table 32).

Table 32. Estimated western North Pacific gray whale exposures to SURTASS LFA sonar at RLs between 120 and 180 dB SPL during the 2016to 2017 annual period.

Site Number	Operating Area	Number of Missions	Marine Mammal Stock	Number in Stock	Percent Stock Affected 120 to 180 dB[1]	Estimated Number of Instances of Behavioral Harassment[2]	Estimated Exposures without Criteria Processing[3]
5	Sea of Japan	1	WNP	140	0.03	1	23
6	East China Sea	1	WNP	140	0.11	1	9
7	South China Sea	1	WNP	140	0.03	1	25
15	Northeast of Japan	0.5	WNP	140	<0.01	1	2
Total Western North Pacific Stock					0.17	4	59
	Totals	**3.5**			**0.17**	**4**	**59**

[1] Percent stock has been rounded up to two decimal points.

[2] This column represents the estimated number of animals that will exhibit a behavioral response (i.e., be affected) by exposure to SURTASS LFA sonar between 120 and 180 dB SPL during the annual period. The SURTASS LFA sonar risk continuum, as described in Section 3.8.1.4, was used to determine how many exposures between 120 and 180 dB SPL would be expected to result in a behavioral response.

[3] This column represents the estimated number of exposures that will occur between 120 and 180 dB SPL (rms) during the 2016 to 2017 annual period. Fractional animals wpotentially affected have been rounded up to the next whole number.

6.4.5.4 Humpback Whale

Humpback whales may be exposed to and affected by SURTASS LFA sonar during a proposed 17 missions in 12 mission areas from August 2016 to August 2017 (Table 33).

Table 33. Estimated humpback whale exposures to SURTASS LFA sonar at RLs between 120 and 180 dB SEL during the 2016 to 2017 annual period.

Site Number	Operating Area	Number of Missions	Marine Mammal Stock or DPS	Number in Stock or DPS	Percent Stock Affected 120 to 180 dB[1]	Estimated Number of Instances of Behavioral Harassment[2]	Estimated Exposures without Criteria Processing[3]
1	East of Japan	1.5	WNP	1,328	0.42	6	2,272
2	North Philippine Sea	3	WNP	1,328	3.27	45	5,346
3	West Philippine Sea	3	WNP	1,328	4.94	67	8,740
4	Offshore Guam	3	WNP	1,328	0.72	11	1,064
7	South China Sea	1	WNP	1,328	0.43	5	2,255
8	Offshore North Pacific (25 to 40° North) (Offshore Japan) 1	1	WNP	1,328	0.11	2	2,889
9	Offshore North Pacific (10 to 25°North) (Offshore Japan)	1	WNP	1,328	0.64	9	2,826
15	Northeast of Japan	0.5	WNP	1,328	0.62	9	2,908
Total Western North Pacific Stock or DPS					11,15	155	28,300
10	Hawaii North (25° North, 158° West)	1	CNP Hawaii	10,103	0.12	13	3,091
11	Hawaii South (19.5° North 158.5° West)	1	CNP Hawaii	10,103	0.14	14	515

Total Central North Pacific Hawaii Stock or DPS					0.26%	27	3,606
12	Arabian Sea	0.5	Arabian Sea	200	0.01	1	16
Total Arabian Sea Stock or DPS					0.01	1	16
14	Northwest of Australia	0.5	WAU	14,000	<0.01	1	43
Total Western Australia Stock or DPS					<0.01	1	43
	Totals	**17**			**11.42**	**184**	**31,965**

[1] Percent stock has been rounded up to two decimal points.

[2] This column represents the estimated number of animals that will exhibit a behavioral response (i.e., be affected) by exposure to SURTASS LFA sonar between 120 and 180 dB SPL during the annual period. The SURTASS LFA sonar risk continuum, as described in Section 3.8.1.4, was used to determine how many exposures between 120 and 180 dB SPL would be expected to result in a behavioral response.

[3] This column represents the estimated number of exposures that will occur between 120 and 180 dB SPL (rms) during the 2016 to 2017 annual period. Fractional animals wpotentially affected have been rounded up to the next whole number.

6.4.5.5 North Pacific Right Whale

North Pacific right whales may be exposed to and affected by SURTASS LFA sonar during a proposed eight missions in six mission areas from August 2016 to August 2017 (Table 34).

Table 34. Estimated North Pacific right whale exposures to SURTASS LFA sonar at RLs between 120 and 180 dB SPL during the 2016 to 2017 annual period.

Site number	Operating Area	Number of Missions	Marine Mammal Stock	Number in Stock	Percent Stock Affected 120 to 180 dB[1]	Estimated Number of Instances of Behavioral Harassment[2]	Estimated Exposures without Criteria Processing[3]
1	East of Japan	1.5	WNP	922	<0.01	1	12
2	North Philippine Sea	3	WNP	922	0.05	2	60
5	Sea of Japan	1	WNP	922	N/A	N/A	0
6	East China Sea	1	WNP	922	<0.02	1	9
7	South China Sea	1	WNP	922	0.03	1	26
15	Northeast of Japan	0.5	WNP	922	0.11	2	33
Total Western North Pacific Stock					0.22	7	140

	Totals	**8**			**0.22**	**7**	**140**

[1] Percent stock has been rounded up to two decimal points.[2] This column represents the estimated number of animals that will exhibit a behavioral response (i.e., be affected) by exposure to SURTASS LFA sonar between 120 and 180 dB SPL during the annual period. The SURTASS LFA sonar risk continuum, as described in Section 3.8.1.4, was used to determine how many exposures between 120 and 180 dB SPL would be expected to result in a behavioral response.

[3] This column represents the estimated number of exposures that will occur between 120 and 180 dB SPL (rms) during the 2016 to 2017 annual period. Fractional animals wpotentially affected have been rounded up to the next whole number.

[4] Not expected in the mission area during that season.

6.4.5.6 Sei Whale

Sei whales may be exposed to and affected by SURTASS LFA sonar during a proposed nine missions in seven mission areas from August 2016 to August 2017 (Table 35).

Table 35. Estimated sei whale exposures to SURTASS LFA sonar at RLs between 120 and 180 dB SPL during the 2016 to 2017 annual period.

Site Number	Operating Area	Number of Missions	Marine Mammal Stock	Number in Stock	Percent Stock Affected 120 to 180 dB[1]	Estimated Number of Instances of Behavioral Harassment[2]	Estimated Exposures without Criteria Processing[3]
1	East of Japan	1.5	NP	7,000	0.22	17	2,285
4	Offshore Guam	3	NP	7,000	0.07	6	555
8	Offshore North Pacific (25 to 40° North) (Offshore Japan)	1	NP	7,000	0.09	7	935
9	Offshore North Pacific (10 to 25° North) (Offshore Japan)	1	NP	7,000	0.98	69	9,206
15	Northeast of Japan	0.5	NP	7,000	0.45	32	976
Total North Pacific Stock					1.81	131	13,957
10	Hawaii North (25° North, 158°	1	Hawaii	178	1.29	3	360

	West)						
11	Hawaii South (19.5° North 158.5° West)	1	Hawaii	178	0.91	2	62
Total Hawaii Stock					2.20	5	422
	Totals	**9**			4.01	**136**	**14,379**

[1] Percent stock has been rounded up to two decimal points.

[2] This column represents the estimated number of animals that will exhibit a behavioral response (i.e., be affected) by exposure to SURTASS LFA sonar between 120 and 180 dB SPL during the annual period. The SURTASS LFA sonar risk continuum, as described in Section 3.8.1.4, was used to determine how many exposures between 120 and 180 dB SPL would be expected to result in a behavioral response.

[3] This column represents the estimated number of exposures that will occur between 120 and 180 dB SPL (rms) during the 2016 to 2017 annual period. Fractional animals wpotentially affected have been rounded up to the next whole number.

6.4.5.7 Sperm Whale

Sperm whales may be exposed to and affected by SURTASS LFA sonar during a proposed 20 nominal missions in 15 mission areas from August 2016 to August 2017 (Table 36).

Table 36. Estimated sperm whale exposures to SURTASS LFA sonar at RLs between 120 and 180 dB SPL during the 2016 to 2017 annual period.

Site Number	Operating Area	Number of Missions	Marine Mammal Stock	Number in Stock	Percent Stock Affected 120 to 180 dB[1]	Estimated Number of Instances of Behavioral Harassment[2]	Estimated Exposures without Criteria Processing[3]
1	East of Japan	1.5	NP	102,112	0.02	20	5,1364
2	North Philippine Sea	3	NP	102,112	0.06	48	10,966
3	West Philippine Sea	3	NP	102,112	0.03	46	11,655
4	Offshore Guam	3	NP	102,112	0.03	36	3,880
5	Sea of Japan	1	NP	102,112	0.03	31	3,004
6	East China Sea	1	NP	102,112	0.01	15	793
7	South China Sea	1	NP	102,112	0.01	11	2,975

8	Offshore North Pacific (25 to 40° North) (Offshore Japan)	1	NP	102,112	0.03	26	8,214
9	Offshore North Pacific (10 to 25° North) (Offshore Japan)	1	NP	102,112	0.02	24	7,037
15	Northeast of Japan	0.5	NP	102,112	0.07	72	6,854
Total North Pacific Stock					0.31	60,742	329
10	Hawaii North (25° North, 158° West)	1	Hawaii	3,354	0.50	17	5,065
11	Hawaii South (19.5° North 158.5° West)	1	Hawaii	3,354	0.42	15	832
Total Hawaii Stock					0.92	32	5,897
12	Arabian Sea	0.5	North Indian	24,446	0.05	13	4,383
13	Andaman Sea	1	North Indian	24,446	0.05	12	456
Total Northern Indian Stock					0.10	25	4,389
14	Northwest of Australia	0.5	South Indian	24,446	0.02	6	590
Total Southern Indian Stock					0.02	6	590
	Totals	**20**			1.35	**392**	**72,068**

[1] Percent stock has been rounded up to two decimal points.

[2] This column represents the estimated number of animals that will exhibit a behavioral response (i.e., be affected) by exposure to SURTASS LFA sonar between 120 and 180 dB SPL during the annual period. The SURTASS LFA sonar risk continuum, as described in Section 3.8.1.4, was used to determine how many exposures between 120 and 180 dB SPL would be expected to result in a behavioral response.

[3] This column represents the estimated number of exposures that will occur between 120 and 180 dB SPL (rms) during the 2016 to 2017 annual period. Fractional animals wpotentially affected have been rounded up to the next whole number.

6.4.5.8 Main Hawaiian Islands Insular DPS of False Killer Whale

The MHI Insular DPS of false killer whales may be exposed to and affected by SURTASS LFA sonar during a proposed two missions in two mission areas from August 2016 to August 2017 (Table 37).

Table 37. Estimated MHI Insular false killer whale exposures to SURTASS LFA sonar at RLs between 120 and 180 dB SPL during the 2016 to 2017 annual period.

Site Number	Operating Area	Number of Missions	Marine Mammal Stock or DPS	Number in Stock or DPS	Percent Stock Affected 120 to 180 dB[1]	Estimated Number of Instances of Behavioral Harassment[2]	Estimated Exposures without Criteria Processing[3]
10	Hawaii North (25° North, 158° West)	1	MHI Insular DPS	151	1.27	1	4,424
11	Hawaii South (19.5° North 158.5° West)	1	MHI Insular DPS	151	0.05	1	748
Total MHI Insular Stock or DPS					1.32	3	5,172
	Totals	**2**			**1.32**	**3**	**5,172**

[1] Percent stock has been rounded up to two decimal points.

[2] This column represents the estimated number of animals that will exhibit a behavioral response (i.e., be affected) by exposure to SURTASS LFA sonar between 120 and 180 dB SPL during the annual period. The SURTASS LFA sonar risk continuum, as described in Section 3.8.1.4, was used to determine how many exposures between 120 and 180 dB SPL would be expected to result in a behavioral response.

[3] This column represents the estimated number of exposures that will occur between 120 and 180 dB SPL (rms) during the 2016 to 2017 annual period. Fractional animals wpotentially affected have been rounded up to the next whole number.

6.4.5.9 Hawaiian Monk Seal

Hawaiian monk seals may be exposed to and affected by SURTASS LFA sonar during a proposed three missions in three mission areas from August 2016 to August 2017 (Table 38).

Table 38. Estimated Hawaiian monk seal exposures to SURTASS LFA sonar at RLs between 120 and 180 dB SEL during the 2016 to 2017 annual period.

Site Number	Operating Area	Number of Missions	Marine Mammal Stock	Number in Stock	Percent Stock Affected 120 to 180 dB[1]	Estimated Number of Instances of Behavioral Harassment[2]	Estimated Exposures without Criteria Processing[3]
8	Offshore North Pacific (25 to 40° North) (Offshore Japan)	1	Hawaii	1,153	0.26	4	400
10	Hawaii North (25° North, 158° West)	1	Hawaii	1,153	0.11	2	368
11	Hawaii South (19.5° North 158.5° West)	1	Hawaii	1,153	0.12	2	66
Total Hawaii Stock					0.49	8	834
	Totals	**3**			**0.49**	**8**	**834**

[1] Percent stock has been rounded up to two decimal points.

[2] This column represents the estimated number of animals that will exhibit a behavioral response (i.e., be affected) by exposure to SURTASS LFA sonar between 120 and 180 dB SPL during the annual period. The SURTASS LFA sonar risk continuum, as described in Section 3.8.1.4, was used to determine how many exposures between 120 and 180 dB SPL would be expected to result in a behavioral response.

[3] This column represents the estimated number of exposures that will occur between 120 and 180 dB SPL (rms) during the 2016 to 2017 annual period. Fractional animals wpotentially affected have been rounded up to the next whole number.

6.4.5.10 *Southern DPS of Spotted Seal*

Southern DPS Spotted seals may be exposed to and affected by SURTASS LFA sonar during a proposed two missions in two mission areas from August 2016 to August 2017 (Table 39).

Table 39. Estimated spotted seal exposures to SURTASS LFA sonar at RLs between 120 and 180 dB SPL during the 2015 to 2016 annual period.

Site Number	Operating Area	Number of Missions	Marine Mammal Stock or DPS	Number in Stock or DPS	Percent Stock Affected 120 to 180 dB[1]	Estimated Number of Instances of Behavioral Harassment[2]	Estimated Exposures without Criteria Processing[3]
5	Sea of Japan	1	Southern	1,000	<0.01	1	25
6	East China Sea	1	Southern	1,000	0.02	1	9
Total Southern Stock or DPS					0.02	2	9
	Totals	**2**			**0.02**	**2**	**34**

[1] Percent stock has been rounded up to two decimal points.

[2] This column represents the estimated number of animals that will exhibit a behavioral response (i.e., be affected) by exposure to SURTASS LFA sonar between 120 and 180 dB SPL during the annual period. The SURTASS LFA sonar risk continuum, as described in Section 3.8.1.4, was used to determine how many exposures between 120 and 180 dB SPL would be expected to result in a behavioral response.

[3] This column represents the estimated number of exposures that will occur between 120 and 180 dB SPL (rms) during the 2016 to 2017 annual period. Fractional animals wpotentially affected have been rounded up to the next whole number

6.4.5.11 Western DPS of Steller Sea Lion

Western DPS Steller sea lions may be exposed to and affected by SURTASS LFA sonar during a proposed 0.5 mission in one mission area from August 2016 to August 2017 (Table 40).

Table 40. Estimated Steller sea lion exposures to SURTASS LFA sonar at RLs between 120 and 180 dB SPL during the 2016 to 2017 annual period.

Site Number	Operating Area	Number of Missions	Marine Mammal Stock or DPS	Number in Stock or DPS	Percent Stock Affected 120 to 180 dB	Estimated Exposures without Criteria Processing	Estimated Number of Instances of Behavioral Harassment
15	Northeast of Japan	0.5	Western	82,516	<0.01	1	34
Total Western Stock or DPS					<0.01	1	34
	Totals	**0.5**			**<0.01**	**1**	**34**

[1] Percent stock has been rounded up to two decimal points.

[2] This column represents the estimated number of animals that will exhibit a behavioral response (i.e., be affected) by exposure to SURTASS LFA sonar between 120 and 180 dB SPL during the annual period. The SURTASS LFA sonar risk continuum, as described in Section 3.8.1.4, was used to determine how many exposures between 120 and 180 dB SPL would be expected to result in a behavioral response.

[3] This column represents the estimated number of exposures that will occur between 120 and 180 dB SPL (rms) during the 2016 to 2017 annual period. Fractional animals wpotentially affected have been rounded up to the next whole number.

6.4.5.12 Sea Turtles

The Navy did not simulate potential exposure of sea turtles to SURTASS LFA sonar transmissions. Due to the seasonal and life stage changes in habitat occupation, sea turtle populations are particularly difficult to census. Abundance estimates are based on the most current information available regarding counts of the most accessible member of the population, nesting females, which does not account for the abundance of male sea turtles. The lack of detailed population data and scarcity of density data for sea turtles at sea do not allow for density estimates to be derived for populations in the North Pacific or Indian Oceans.

Juvenile and adult stages of sea turtles could potentially be exposed to SURTASS LFA sonar transmissions when residing in pelagic environments. We assume that the monitoring protocols associated with SURTASS LFA sonar would be more effective with larger sea turtles, such as adult leatherback and loggerhead turtles, than with smaller species such as olive ridley, hawksbill, or green turtles or smaller individual leatherback or loggerhead turtles. Monitoring measures may not detect some individual or smaller sea turtles, which would increase their risk of exposure to sound pressure levels associated with SURTASS LFA sonar within the mitigation zone (that is, 180 dB) if they encountered SURTASS LFA sonar vessels during sonar transmission.

Due to the small size of the LFA mitigation zone relative to the enormous area and volume of the ocean and the water column usage by sea turtles (i.e., generally closer to the surface in open water habitats), and the three-part monitoring system, sea turtles are unlikely to be exposed to SURTASS LFA sonar transmission levels greater than or equal to 180 dB re: 1 µPa (rms), which is the threshold levels above which they may be injured. For instance, in 2011, a sea turtle was observed by visual monitors aboard the USNS *EFFECTIVE* at a distance of about 15 m (49 ft) from the vessel over eight minutes after LFA sonar transmissions ceased. Due to the proximity to the vessel, an assessment was made to determine the potential for the sea turtle to be within the LFA mitigation zone (180 dB sound field) and to determine whether the HF/M3 sonar should have detected the turtle. The assessment concluded that due to the position of the HF/M3 sonar system above the LFA VLA, the sea turtle would have had to swim from the surface through the HF/M3 sonar detection zone to enter into the 180-dB LFA mitigation zone, making the potential for an acoustic detection of the turtle highly likely. The lack of detection by the HF/M3 sonar would indicate that the sea turtle remained at or near the surface and did not dive into the 180 dB LFA mitigation zone during transmissions. Thus, it is improbable that the sea turtle received SPL from SURTASS LFA sonar at or above 180 dB re: 1 µPa (rms). It is more probable that sea turtles could be exposed to SURTASS LFA sonar at levels less than 180 dB re: 1 µPa (rms), which are levels that could cause a behavioral response.

6.5 Response Analysis

The Risk Assessment Framework for SURTASS LFA sonar (Section 6.2) discussed the potential responses of mysticetes, odontocetes, and pinnipeds to exposure to SURTASS LFA sonar. Based on the literature presented in these sections and the exposure estimates presented in Section 6.4.5, this section details our conclusions on the response of mysticetes, odontocetes, and pinnipeds to exposure to this stressor.

Because of the many unknowns about the relationships between an animal's behavioral responses and their population ecology, the U.S. Navy and NMFS have interpreted any data or other information conservatively when dealing with the SURTASS LFA sonar program. As a result, when an animal's responses to LFA sonar transmissions appear to be a low-level response to a novel stimulus (such as an alert response or a startle response), or a response with unknown consequence, the U.S. Navy and NMFS has chosen to treat these responses as symptomatic of potential disruptions of one or more behavioral patterns including, but not limited to, temporary changes in vocalizations or dive patterns, temporary avoidance of an area, or temporary disruption of feeding, migrating, or reproductive behaviors. The estimates calculated (percent of stock affected at less than 180 dB SPL) 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 considered the available scientific evidence to determine the likely nature of the modeled marine mammal behavioral responses and the potential fitness consequences for affected individuals.

6.5.1 Mysticetes

As discussed previously, based on the performance of the SURTASS LFA sonar system and the mitigative measures the U.S. Navy has used with the sonar system over the past 12 years (2002 to 2012), we do not expect any baleen whales (blue, fin, Western North Pacific gray, humpback, North Pacific right, or sei whales) to be exposed to received levels equal to or greater than 180 dB. The available evidence suggests that at received levels below 180 dB SEL, exposure to LFA sonar transmissions are not likely to result in injury, masking, stranding, resonance effects, or other long-term behavioral effects in marine mammals. Many animals exposed at these received levels may not respond at all. For example, 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). The best scientific and commercial data available (presented in Sections 6.2.3 and 6.5) suggest that exposing individual mysticetes to SURTASS LFA sonar may cause those animals to react behaviorally. For example, as part of determining the potential effects of SURTASS LFA sonar, the U.S. Navy conducted the LFS SRP. In response to LFA sonar, blue and fin whales did not exhibit any overt behavioral responses or changes in distribution, whereas gray whales avoided the sound source following exposure and humpback whales ceased singing during sonar transmissions. All responses to the sound source were short-term (Clark and Fristrup 2001; Croll et al. 2001b; Fristrup et al. 2003; Miller et al.

2000b; Nowacek et al. 2007). These temporary responses (most likely lasting only a few minutes) are not expected to impact the fitness of any individual animals as the responses are not likely to adversely affect the ability of the animals to forage, detect predators, select a mate, or reproduce successfully. Because we do not expect any fitness consequences from any individual animals, we do not expect any population (stock) level effects from these behavioral responses.

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). 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. 2012b; Goldbogen et al. 2013; Kight and Swaddle 2011; McGregor 2013; Read et al. 2014b; Williams et al. 2014b). However, although there is significant uncertainty, based upon the available evidence and the foregoing analysis, we conclude that modeled instances of behavioral disturbance is unlikely to cause aggregate or long-term adverse effects on the baleen whales considered in this opinion and conference report, such as abandonment of important habitat or adverse physiological effects resulting from chronic or long-term stress.

6.5.2 Odontocetes

Sperm and MHI insular false killer whales are considered mid-frequency cetaceans, with functional hearing conservatively estimated to be between approximately 150 Hz and 160 kHz (Southall et al. 2007a). Therefore, these species are likely to be able to detect LFA sonar. As discussed previously, based on the performance of the SURTASS LFA sonar system and the mitigative measures the U.S. Navy has used with the sonar system over more than 12 years of its use (2002 to 2015), we do not expect any odontocetes (sperm whales and MHI insular false killer whales) to be exposed to received levels equal to or greater than 180 dB SEL. Similar to mysticetes, it’s possible that many animals exposed at these received levels may not respond at all. For example, a study monitored the behavioral responses of sperm whales to a LFA sonar system that was being developed for use by the British Navy and found no evidence of behavioral responses that could be attributed to the LFA sonar (Aicken et al. 2005). The available evidence suggests that at received levels below 180 dB, exposure to SURTASS LFA sonar transmissions are not likely to result in injury, masking, stranding, resonance effects, or other long-term behavioral effects in marine mammals. Though data are limited that consider the behavioral response of odontocetes to low-frequency sources, the best scientific and commercial data available (presented in Section 6.2.3 and 6.5) suggest that exposing individual odontocetes to SURTASS LFA sonar may cause those animals to react behaviorally.

For example, to preserve the saliency of their vocalizations and the coherence of their social interactions, MHI insular false killer whales might have to make one or more vocal adjustments. Because any reductions in the active space of whale vocalizations that result from SURTASS

LFA sonar would be temporary and episodic, any vocal adjustments MHI insular false killer whales would have to make would also be temporary. Studies on a captive false killer whale have demonstrated an ability to proactively change hearing sensitivity, apparently for protection, when a warning sound was provided prior to a more intense sound and that the animal learned to change its hearing sensitivity when warned that a loud sound was about to arrive. (Nachtigall and Supin 2013). Controlled exposure experiments in the Bahamas found that after each MFA reception of simulated MFA sonar sounds (3.1 to 3.4 kHz, source level 160 to 211 dB re: 1 µPa) false killer whales increased their whistle rate and produced whistles that were more similar in their frequency characteristics to MFA sonar (Goldbogen et al. 2013). If necessary, we would expect sperm and MHI false killer whales to be able to similarly alter their communication in response to LFA sonar.

Similar to the behavioral reactions of mysticetes, as discussed above, behavioral responses are expected to be temporary (most likely lasting only a few minutes) are not expected to impact the fitness of any individual animals as the responses are not likely to adversely affect the ability of the animals to forage, detect predators, select a mate, or reproduce successfully. Because we do not expect any fitness consequences to any individual animals, we do not expect any population (stock or DPS) level effects from these behavioral responses.

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). 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. 2012b; Goldbogen et al. 2013; Kight and Swaddle 2011; McGregor 2013; Read et al. 2014b; Williams et al. 2014b). For example, as discussed previously, 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 (also odontocetes). However, 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. Although there is uncertainty, based upon the available evidence and the foregoing analysis, we conclude that modeled instances of behavioral disturbance is unlikely to cause aggregate or long-term adverse effects on the baleen whales considered in this opinion and conference report, such as abandonment of important habitat or adverse physiological effects resulting from chronic or long-term stress.

6.5.3 Pinnipeds

As discussed previously, based on the performance of the SURTASS LFA sonar system and the mitigation measures the U.S. Navy has used with the sonar system over more than 13 years (2002 to 2016) of sonar deployment, we do not expect any pinnipeds (Hawaiian monk seals, spotted seals, or Steller sea lions) to be exposed to received levels equal to or greater than 180 dB SEL. Similar to mysticetes and odontocetes, it's possible that many animals exposed at these received levels may not respond at all. For example, low-frequency signals of the Acoustic Thermometry of Ocean Climate sound source were not found to overtly affect elephant seal dives off California (Costa et al. 2003). The available evidence suggests that at received levels below 180 dB, exposure to LFA sonar transmissions are not likely to result in injury, masking, stranding, resonance effects, or other long-term behavioral effects in pinnipeds. The best scientific and commercial data available (presented in Section 6.2.3 and 6.5) suggest that exposing individual marine mammals to SURTASS LFA sonar may cause those animals to react behaviorally, but those behavioral reactions are not likely to adversely affect the ability of the animals to forage, detect predators, select a mate, or reproduce successfully.

Behavioral responses can range from a mild orienting response, or a shifting of attention, to flight and panic. Research and observations show that pinnipeds in the water are tolerant of anthropogenic noise and activity. They may react in a number of ways depending on their experience with the sound source and what activity they are engaged in at the time of the exposure. For example, for a disruption in foraging to result in significant fitness consequences we would have to assume that an individual animal detects and responds to the acoustic source, and that it could not compensate for lost feeding opportunities by either immediately feeding at another location, by feeding shortly after cessation of acoustic exposure, or by feeding at a later time. There is no indication this is the case, particularly since foraging habitat would still be available in the environment following the cessation of acoustic exposure. Elephant seals (*Mirounga angustirostris*) 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) cited in (New et al. 2014). Similarly, since Hawaiian monk seals, spotted seals, and Steller sea lions have near continuous opportunities to feed throughout the year, they are unlikely to be affected by short-term variations in prey availability. Therefore, behavioral responses of these species to LFA sonar are unlikely to lead to fitness consequences to individual seals or long-term implications for the population (stock).

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). 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. 2012b; Goldbogen et al. 2013; Kight and Swaddle 2011; McGregor 2013; Read et al. 2014b; Williams et al. 2014b). 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 ten percent.

The findings of New et al. (2014) are not applicable to the temporary behavioral disruptions that may be caused by SURTASS LFA sonar 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 and conference report to be similarly unaffected. We have no information to suggest animals eliciting a behavioral response (e.g., temporary disruption of feeding) from exposure to SURTASS LFA sonar 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 SURTASS LFA sonar will not have as long of a duration as those considered in the New et al. (2014) study. No individual animals will be exposed to SURTASS LFA sonar 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).

Although there is significant uncertainty, based upon the available evidence and the foregoing analysis, we conclude that modeled instances of behavioral disturbance is unlikely to cause aggregate or long-term adverse effects on the marine mammals considered in this opinion and conference report, such as abandonment of important habitat or adverse physiological effects resulting from chronic or long-term stress.

6.5.4 Sea Turtles

As explained in Section 6.4.5.12, due to the lack of more definitive data on sea turtle stock distributions in the open ocean, it is not feasible to estimate the percentage of a stock (or number of individuals) that could be located in a SURTASS LFA sonar operations area at a potentially vulnerable depth, during an LFA sonar sound transmission.

Little is known about how sea turtles (green, hawksbill, loggerhead, olive ridley, leatherback) use sound in their environment. Based on knowledge of their sensory biology (Bartol and Ketten 2006; Moein Bartol and Musick 2003), sea turtles may be able to detect objects within the water column (e.g., vessels, prey, predators) via some combination of auditory and visual cues. However, research examining the ability of sea turtles to avoid collisions with vessels shows they may rely more on their vision than auditory cues (Hazel et al. 2007). Similarly, while sea turtles may rely on acoustic cues to identify nesting beaches, they appear to rely on other non-acoustic cues for navigation, such as magnetic fields (Lohmann and Lohmann 1996a; Lohmann and Lohmann 1996b) and light (Avens and Lohmann 2003). Additionally, they are not known to produce sounds underwater for communication.

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 (less than 2 kHz) (Bartol et al. 1999a; Dow Piniak et al. 2012; Lenhardt et al. 1983; Lenhardt et al. 1994; Martin et al. 2012; O'Hara and Wilcox 1990; Ridgway et al. 1969), with greatest sensitivity below one kHz. A more recent review on sea turtle hearing and sound exposure indicated that sea turtles detect sounds at less than 1,000 Hz (Popper et al. 2014).

No data are available indicating the potential response of sea turtles to sonar (Popper et al. 2014). McCauley et al. (2000) studied the response of green and loggerhead sea turtles to 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. They reported that the sea turtles responded consistently above received levels of about 166 dB re: 1 μPa: they increased their swimming activity compared to periods during which the airgun was not operating. Above 175 dB re: 1 μPa their behavior became erratic and might have indicated an agitated state.

O’Hara and Wilcox (1990) exposed loggerhead sea turtles to low-frequency sound from two types of seismic airguns with source levels approximating 256 and 262 dB re: 1μPa (the paper does not state whether this was peak, peak-to-peak, or root mean square). With airguns firing at a rate of four times per minute and source levels of 256 dB, the distribution of the loggerhead sea turtles was not significantly different than during control trials. When they increased source levels to about 262 dB, the sea turtles avoided the seismic airgun when it fired at four or eight times per minute. Although the sound source in this experiment is not directly comparable to the low-frequency active sonar associated with the proposed operations of the SURTASS LFA sonar system, the results of this study illustrate the relative insensitivity of sea turtles (at least loggerhead sea turtles) to low-frequency sound.

In their *Sound Exposure Guidelines for Sea Turtles and Fishes*, a technical report developed and approved by Accredited Standards Committee S3/SC 1 Animal Bioacoustics, Popper et al. (2014) determined that the relative risk of sea turtles responding behaviorally to low-frequency sonar is low, regardless of the distance from the source. Because of this, we do not expect sea turtles would exhibit long term behavioral responses that could impact an individual's fitness if they were exposed to SURTASS LFA sonar. Popper et al. (2014) also concluded that the risk of sea turtles experiencing immediate mortality, serious injury that could lead to death, or recoverable injury was low from exposure to low-frequency sonar. Further, if the animal was located far from the sound source, there was a low risk it could experience TTS, and a moderate risk if the animal was located intermediate or close distances to the source. Due to the monitoring and mitigation measures implemented during SURTASS LFA sonar transmissions that would reduce the likelihood of sea turtles being in close proximity to the sound source, any exposure of sea turtles to LFA sonar signals is not expected to result in mortality, injury, or TTS.

Based on the limited evidence available, sea turtles exposed to LFA sonar may hear the sound. At levels below 175 dB re: 1 μPa, they may not react at all or they may increase their swimming speed. Any changes in their swimming behavior would not be expected to result in a measurable risk to individual sea turtles.

6.6 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.

Most of the action area is outside of the territorial waters of the United States of America. We are not aware of any state or private activities that are likely to occur in the action area during the foreseeable future that were not considered in the *Environmental Baseline* section of this opinion and conference report.

6.7 Repeated Exposures and Potential for Long-Term or Additive Impacts

To address a Court ruling that we take a long-term view of the "action" (*Natural Resources Defense Council et al. v. Pritzker et al.*, 62 F.Supp.3d 969; United States District Court, N.D. California, 2014), we assumed for purposes of this opinion and conference report that the U.S. Navy's SURTASS LFA sonar routine training, testing, and military operational activities and associated impacts will continue into the reasonably foreseeable future at the levels assessed in this opinion and conference report. This assumption raises the question of whether the U.S. Navy's activities are reasonably certain to cause any aggregate or long-term impacts over time, beyond the effects of individual takes that could occur in a given year. Further information is provided below.

To address the likelihood of long-term additive or accumulative impacts, 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.

Regarding stressors that accumulate in the environment, 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[11].

For effects representing the individual, population, or species' response to the accumulation of stressors, 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; 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 immigrate or emigrate from 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.

We concluded that the effects of repeated exposures to SURTASS LFA sonar were not likely to accumulate through altered energy budgets caused by avoidance behavior (reducing the amount of time available to forage), physiological stress responses (mobilizing glucocorticosteroids, which increases an animal's energy demand), or the canonical costs of changing behavioral states (small decrements in the current and expected reproductive success of individuals exposed to the stressors). In particular, we concluded that the species would be exposed on foraging areas and would experience trivial increases in feeding duration, effectiveness, or both, that would not accumulate in a manner that is likely to result in avoidance behavior or altered energy budgets. In short, the vast majority of impacts expected from sonar exposure are behavioral in nature, temporary and comparatively short in duration, relatively infrequent, and not of the type or

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[11] These example phenomena are not a concern in this opinion and conference report.

severity that would be expected to be additive for the small portion of the stocks and species likely to be exposed either annually, or in the reasonably foreseeable future.

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). However, the U.S. Navy's routine testing, training, and military operations involving SURTASS LFA sonar are infrequent and short-term.. Even though an animal's exposure to LFA 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 U.S. Navy's SURTASS LFA sonar routine testing, training, and military operations do not create conditions of chronic, continuous underwater noise, and will not lead to habitat abandonment or long-term hormonal or physiological stress responses in marine mammals.

As documented above, impacts from sonar exposure are expected to be behavioral in nature, temporary and comparatively short in duration, relatively infrequent, and not of the type or severity that would be expected to be additive for the small portion of the stocks and species likely to be exposed annually, into the reasonably foreseeable future. Thus, while the number of individuals "taken" by active sonar increases over time, the effect of each "take" on the survival or reproductive success of the animals themselves would not accumulate in the same way. As a result, for example, we do not expect that exposing one whale to LFA sonar per year at a level that we would consider a take in the form of behavioral harassment, as predicted by the U.S. Navy's modeling described above, would result in effects over the long-term that would be greater than what we would expect from a single exposure event. To the contrary, we do not expect the effects of that "take" to have any additive, interactive, or synergistic effect on the individual animals, the population(s) those individuals represent, or the species those population(s) comprise.

The preliminary findings from Melcon et al. (2012) and Goldbogen et al. (2013), are also consistent with our determination that behavioral responses of mysticetes to active sonar and other active acoustic sources are unlikely to have any measurable adverse impact on the long-term fitness or reproductive success of individual animals or long-term adverse population-level effects. Although Goldbogen et al. (2013) speculates that "frequent exposures to mid-frequency anthropogenic sounds may pose significant risk to the recovery rates of endangered blue whale populations," the authors acknowledge that the actual responses of individual blue whales to simulated mid-frequency sonar documented in the study "typically involves temporary avoidance responses that appear to abate quickly after sound exposure." Moreover, the most significant response documented in the study occurred not as a result of exposure to simulated mid-frequency sonar but as a result of exposure to pseudo-random noise. Therefore, the overall weight of scientific evidence indicates that substantive behavioral responses by mysticetes, if any, from exposure to LFA sonar and other active acoustic sources evaluated in this opinion and conference report are likely to be temporary and are unlikely to have any long-term adverse impact on individual animals or affected populations.

As discussed above in Section 6.2.7, recent 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.

With respect to threatened and endangered marine mammals, our conclusion that the annual predicted behavioral takes resulting from exposure to SURTASS LFA sonar continuing into the reasonably foreseeable future, are unlikely to result in accumulated adverse impacts is consistent with the negligible impact determination contained in the MMPA rulemaking, which is incorporated by reference. See 75 FR 69317-18; 74 FR 33828, 33884-92.

Our assessment that the continuation of the U.S. Navy's SURTASS LFA sonar routine training, testing, and military operational 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 U.S. Navy activities to date, despite more than a decade of U.S. Navy SURTASS LFA sonar use. Most of the SURTASS LFA sonar routine training, testing, and military operational activities are similar, if not identical, to activities that have been occurring in the same locations for decades.

The best assessment of long-term consequences from SURTASS LFA sonar routine training, testing, and military operations will be to monitor the populations over time within the action area. A U.S. 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.

7 INTEGRATION AND SYNTHESIS

Our effects analyses identified the probable risks to ESA-listed species associated with U.S. Navy routine training, testing, and military operation of SURTASS LFA sonar in the specified areas described in the U.S. Navy's application for LOAs, and NMFS' issuance of an annual LOA to authorize take of marine mammals from the period of August 15, 2016 through August 14, 2017. We measure risks to individuals of endangered or threatened species using changes in the individuals' "fitness" or the individual's growth, survival, annual reproductive success, and lifetime reproductive success. When we do not expect ESA-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 ESA-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.

Routine training, testing, and military operation of SURTASS LFA sonar in the action area will continue to introduce a suite of stressors into the marine and some coastal ecosystems. We determined that transmission of LFA sonar would likely adversely affect ESA-listed species while other stressors associated with these activities such as surface vessel movement, high-frequency active sonar and visual cues from surface vessels as they move through the ocean’s surface would not result in responses that rise to the level of take, although remain part of the overall context of exposure to LFA sonar. Depending on the level of exposure and severity of the response, repeatedly exposing endangered and threatened marine animals to LFA sonar could pose additional risks. Repeated exposures are discussed in more detail in section 6.7. More importantly, endangered and threatened marine animals that occur in the action area would be exposed to combinations of stressors produced by U.S. Navy SURTASS LFA sonar activities at the same time they are exposed to stressors from other human activities and natural phenomena that form the *Environmental Baseline*. 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 and conference report, we assume that routine training, testing, and military operation of SURTASS LFA sonar in the action area and associated effects to ESA-listed species will continue into the reasonably foreseeable future at the levels proposed by the U.S. Navy as described in this opinion and conference report and in the MMPA LOAs. 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. Repeated exposure of individuals to acoustic stress at high received levels have the potential to cause auditory fatigue and hearing loss. However, as described throughout this opinion and conference report, we do not expect animals to be exposed to SURTASS LFA sonar at received levels that could cause auditory fatigue or hearing loss (TTS PTS), largely because of the mitigation and monitoring measures the U.S. Navy employs to minimize the likelihood of such exposures. Further, U.S. Navy routine training, testing, and military operations in the action area involving SURTASS LFA sonar are infrequent and short-term in nature. Even though an animal may be exposed to SURTASS LFA sonar more than one time, the intermittent nature of the sonar signal, its 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. If individual animals are exposed to SURTASS LFA sonar multiple times, we expect these individuals to be able to recover between exposures because of the intermittent nature and duration of SURTASS LFA sonar. U.S. Navy routine training, testing, and military operations involving SURTASS LFA sonar 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 the ESA-listed species considered in this opinion and conference report.

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), 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). Given these animal's mobility and large ranges, we would expect these individuals to temporarily select alternative habitat nearby until the exposure levels in their initially selected area have decreased. Therefore, 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. Because we do not expect any fitness consequences from any individual animals, we do not expect any population level effects from these behavioral responses.

Further, establishing a causal link between anthropogenic noise, animal communication, and individual impacts as well as population viability is difficult to quantify and assess (McGregor 2013) (Read et al. 2014b). The difficulty in assessing the effects of sounds individually and cumulatively on marine species is the confounding nature of indirect effects, age class, prior experience, behavioral state at the time of exposure, and that responses may be influenced by other non-sound related factors (Ellison et al. 2012b; Goldbogen et al. 2013; Kight and Swaddle 2011; McGregor 2013; Williams et al. 2014b) (Read et al. 2014b). Though there is uncertainty, based upon the available evidence and the foregoing analysis, we conclude that modeled instances of behavioral disturbance is unlikely to cause aggregate or long-term adverse effects on the species considered in this opinion and conference report, such as abandonment of important habitat or adverse physiological effects resulting from chronic or long-term stress.

Our assessment that the continuation of the U.S. Navy's SURTASS LFA sonar routine training, testing, and military operational 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 these activities to date in our ESA section 7 consultation history. Most of the SURTASS LFA sonar routine training, testing, and military operations the U.S. Navy conducts in the action area are similar, if not identical, to activities that have been occurring in the same locations for several years.

Our regulations require us to consider, using the best available scientific data, effects of the action that are “likely” and “reasonably certain” to occur rather than effects that are speculative or uncertain. See 50 C.F.R. § 402.02 (defining to “jeopardize the continued existence of” and “effects of the action”). Our analysis and conclusions in this opinion and conference report are based on estimates of exposures and take assuming that the U.S. Navy conducts the maximum extent of SURTASS LFA sonar routine training, testing, and military operational activities as assessed in this opinion and conference report, and the MMPA incidental take authorization. The effects of the action in relation to the *Status of Listed Resources* and the *Environmental Baseline* are presented for each species in the following sections:

7.1 Blue Whale

In determining whether U.S. Navy SURTASS LFA sonar routine training, testing, and military operational activities in the 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. SURTASS LFA sonar activities in the action area proposed from August 15, 2016 through August 14, 2017 are expected to continue at similar levels into the reasonably foreseeable future. Many of these activities will occur without any blue whales being exposed to sound fields associated with SURTASS LFA sonar. 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 and conference report, 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 virtually eliminated in the North Pacific Ocean. The current abundance trend for blue whales rangewide including the 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).

As described in section 6.4.5 of this opinion and conference report, assuming the U.S. Navy conducts the maximum extent of routine training, testing, and military operations in a year, we estimated there will be approximately 16 instances of blue whale behavioral harassment from SURTASS LFA sonar. The individuals affected would be from the Western North Pacific, the Central North Pacific – Hawaii, North Indian, and South Indian stocks of blue whales, which are estimated at 9,250, 81, 3,442, and 1,657 blue whales, respectively. The highest number of instances would occur in the West Philippine Sea mission areas. The 16 instances of behavioral harassment would result in 0.02 percent of the Western North Pacific stock, 1.53 percent of the Central North Pacific – Hawaii stock, 0.01 of the North Indian stock, and 0.03 of the South Indian stock being affected by SURTASS LFA sonar transmissions during 16 missions.

The evidence available suggests that when exposed to SURTASS LFA sonar transmissions, blue whales, like other baleen whales, are not likely to be killed or experience injury, masking, stranding, resonance effects, or behavioral responses that might reduce the longevity or reproductive success of individuals that have been exposed. Instead, the best scientific and commercial data available suggests that SURTASS LFA sonar transmissions are likely to elicit short-term behavioral responses in blue whales that are not known to have long-term, adverse consequences for the biology or ecology of the individual whales exposed to an LFA sonar signal.

We do not anticipate any take in the form of injury from PTS or other injuries such as gastrointestinal tract or lung injury during routine training, testing, or military operational activities. We do not anticipate any mortality of blue whales from SURTASS LFA sonar stressors. The estimates of exposures to LFA sonar signals that would result in behavioral responses 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.

Goldbogen et al. (2013) speculated that if this temporary behavioral response interrupted feeding behavior, this could have impacts on individual fitness and eventually, population health. However, for this to be true, we would have to assume that an individual whale could not compensate for this lost feeding opportunity by either immediately feeding at another location, by feeding shortly after cessation of acoustic exposure, or by feeding at a later time. There is no indication 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.

Most of the blue whales that are exposed to acoustic stressors would be exposed periodically or episodically over certain months or seasons from SURTASS LFA sonar in the action area. Given the nature of SURTASS LFA sonar routine testing, training, and military operations as described above, these periodic or episodic exposure and behavioral response scenarios most often allow sufficient time to return to baseline conditions and resumption of normal activities such as feeding, breeding, or migrating. As described previously in this opinion and conference report, the available scientific information does not provide evidence that exposure to acoustic stressors from SURTASS LFA sonar will impact the fitness of any individuals of this species. Therefore exposure to acoustic stressors will not have population or species level impacts.

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 SURTASS LFA sonar 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 SURTASS LFA sonar activities is not expected to impact the fitness of any individuals of this species. No mortality of blue whales is expected to occur from SURTASS LFA sonar activities.

Based on the evidence available, including the *Environmental Baseline* and *Cumulative Effects*, and stressors resulting from U.S. Navy SURTASS LFA sonar routine training, testing, and military operations that will be conducted in the action area on an annual basis or cumulatively for the reasonably foreseeable future (assuming there are no significant changes to the *Status of Listed Resources* or *Environmental Baseline*), would not be expected, directly or indirectly, 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 U.S. Navy SURTASS LFA sonar routine training, testing, and military operational 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. SURTASS LFA sonar 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. An action that is not likely to reduce the fitness of individual whales is not 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) and therefore, we do not expect acoustic stressors to result in substantial changes in reproduction, numbers, or distribution of these populations. By the same logic, we do not anticipate any reductions in survival rate or trajectory of recovery of the species as listed pursuant to the ESA that could be readily perceived or estimated.

7.2 Fin Whale

In determining whether U.S. Navy SURTASS LFA sonar routine training, testing and military operational activities in the 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. SURTASS LFA sonar activities in the action area proposed from August 15, 2016, through August 14, 2017, are expected to continue at similar levels into the reasonably foreseeable future. Many of these activities will occur without any fin whales being exposed to sound fields associated with LFA sonar. 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 and conference report, some of the primary anthropogenic threats to the survival and recovery of fin whales have been whaling and ship strikes. The threat of whaling has been eliminated. The current abundance trend for fin whales rangewide including the action area is not well understood.

As described in section 6.4.5 of this opinion and conference report, assuming the U.S. Navy conducts the maximum extent of routine training, testing, and military operational activities in a year, we estimated there will be approximately 107 instances of fin whale behavioral harassment from SURTASS LFA sonar. The highest number of instances of behavioral harassment (31) would occur in the Sea of Japan where U.S. Navy proposes to conduct one SURTASS LFA sonar mission. Five fin whale stocks will be affected; the Western North Pacific stock - 85 exposures out of a total stock estimated to include 9,250 fin whales; the East China Sea stock – five instances of behavioral harassment out of a stock estimated at 500 individuals; the Hawaii stock - two instances of behavioral harassment out of a stock estimated at 58 individuals; the Northern Indian stock – three instances of behavioral harassment out of a stock estimated at 1,716 individuals; and the South Indian stock – 13 instances of behavioral harassment out of a stock estimated at 38,185 individuals. The 107 instances of behavioral harassment would result in 0.81 percent of the Western North Pacific stock, 0.75 percent of the East China Sea stock, 1.31 percent of the Hawaii stock, 0.07 of the Northern Indian stock, and 0.03 of the Southern Indian stock being affected by SURTASS LFA sonar transmissions during 20 nominal missions.

The evidence available suggests that fin whales, like other baleen whales, exposed to SURTASS LFA sonar transmissions are not likely to be killed or experience injury, masking, stranding, resonance effects, or behavioral responses that might reduce the longevity or reproductive success of individuals that have been exposed. Instead, the best scientific and commercial data available suggests that SURTASS LFA sonar transmissions are likely to elicit short-term behavioral responses in fin whales that are known to have no long-term, adverse consequences for the biology or ecology of the individual whales exposed to an LFA sonar signal.

We do not anticipate any take in the form of injury from PTS or other injuries such as GI tract or lung injury during SURTASS LFA sonar routine training, testing, or military operational activities. We do not anticipate any mortality of fin whales from SURTASS LFA sonar stressors. The estimates of exposures to LFA sonar signals that would result in behavioral responses 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.

We consider fin whales to be able to hear and respond to low-frequency sonar similar to blue whales. Most of the fin whales that are exposed to acoustic stressors would be exposed periodically or episodically over certain months or seasons from military activities in the action area. Given the nature of SURTASS LFA sonar routine training, testing, and military operations as described above, these periodic or episodic exposure and behavioral response scenarios most often allow sufficient time to return to baseline conditions and resumption of normal behavioral activities such as feeding, breeding, or migrating. As described previously in this opinion and conference report, the available scientific information does not provide evidence that exposure to acoustic stressors from LFA sonar 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 SURTASS LFA sonar activities. As discussed previously, anthropogenic noise associated with SURTASS LFA sonar 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 one percent chance of extinction in 100 years. To qualify for delisting, each recovery population must also have no more than a ten 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 U.S. Navy SURTASS LFA sonar routine training, testing, and military operational activities proposed August 15, 2016 through August 14, 2017 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, directly or indirectly, to appreciably reduce the likelihood of the survival of fin whales in the wild by reducing the reproduction, numbers, or distribution of that species. We also conclude that effects from ongoing SURTASS LFA sonar 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. SURTASS LFA sonar 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 is not 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) and therefore, we do not expect acoustic stressors to result in substantial changes in reproduction, numbers, or distribution of these populations.. By the same logic, we do not anticipate any reductions in survival rate or trajectory of recovery of the species as listed pursuant to the ESA that could be readily perceived or estimated.

7.3 Western North Pacific Gray Whale

In determining whether U.S. Navy SURTASS LFA sonar routine training, testing and military operational activities in the action area are likely to jeopardize the survival and recovery of western North Pacific gray whales, we assessed effects of the action against the aggregate effects of everything in the *Environmental Baseline* that has led to the current *Status of Listed Resources* and, those effects of future non-Federal activities that are reasonably certain to occur within the action area in the reasonably foreseeable future. SURTASS LFA sonar activities in the action area proposed from August 15, 2016, through August 14, 2017, are expected to continue at similar levels into the reasonably foreseeable future. Many of these activities will occur without any western North Pacific gray whales being exposed to sound fields associated with LFA sonar. 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 and conference report, some of the primary anthropogenic threats to the survival and recovery of western North Pacific gray whales have been whaling and ship strikes. The threat of whaling has been eliminated. The current abundance trend for western North Pacific gray whales rangewide including the action area is not well understood.

As described in section 6.4.5 of this opinion and conference report, assuming that the U.S. Navy conducts the maximum extent of routine training, testing, and military operational activities in a year, we estimated there will be approximately four instances of western North Pacific gray whale behavioral harassment from SURTASS LFA sonar. The western North Pacific stock of gray whales is estimated to include only 140 animals. The four instance of behavioral harassment would result in 0.17 percent of the Western North Pacific stock being affected by SURTASS LFA sonar transmissions during four missions.

We do not anticipate any take in the form of injury from PTS or other injuries such as GI tract or lung injury during routine training, testing, or military operations. We do not anticipate any mortality of western North Pacific gray whales from SURTASS LFA sonar stressors. The estimates of exposures to LFA sonar signals that would result in behavioral responses would not be expected to appreciably reduce the likelihood of the survival of western North Pacific gray whales in the wild by reducing the reproduction, numbers, or distribution of that species for the reasons discussed herein.

The evidence available suggests that western North Pacific gray whales, like other baleen whales, exposed to SURTASS LFA sonar transmissions are not likely to be killed or experience injury, masking, stranding, resonance effects, or behavioral responses that might reduce the longevity or reproductive success of individuals that have been exposed. Instead, the best scientific and commercial data available suggests that SURTASS LFA sonar transmissions are likely to elicit short-term behavioral responses that are known to have no long-term, adverse consequences for the biology or ecology of the individual whales exposed to an LFA sonar signal.

The western North Pacific gray whale does not have a recovery plan; therefore, specific downlisting and delisting criteria are not established. We concluded no mortality of individuals would occur and that effects from acoustic stressors would be temporary and not impact the fitness of individuals or the population. In the absence of fitness consequences on individuals or the population to which those individuals belong, we do not expect an appreciable reduction in the ability of the western North Pacific gray whale to survive or recover.

Based on the evidence available, including the *Environmental Baseline* and *Cumulative Effects*, stressors resulting from U.S. Navy SURTASS LFA sonar routine training, testing, and military operational activities proposed August 15, 2016 through August 14, 2017 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, directly or indirectly, to appreciably reduce the likelihood of the survival of western North Pacific gray whales in the wild by reducing the reproduction, numbers, or distribution of that species. We also conclude that effects from ongoing SURTASS LFA sonar activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of western North Pacific gray whales in the wild by reducing the reproduction, numbers, or distribution of that species. SURTASS LFA sonar stressors will not affect the population dynamics, behavioral ecology, and social dynamics of individual gray 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 is not 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) and therefore, we do not expect acoustic stressors to result in substantial changes in reproduction, numbers, or distribution of these populations. By the same logic, we do not anticipate any reductions in survival rate or trajectory of recovery of the species as listed pursuant to the ESA that could be readily perceived or estimated.

7.4 Humpback Whale

In determining whether U.S. Navy SURTASS LFA sonar routine training, testing and military operational activities in the action area are likely to jeopardize the survival and recovery of humpback whales, we assessed effects of the action against the aggregate effects of everything in the *Environmental Baseline* that has led to the current *Status of Listed Resources* and, those effects of future non-Federal activities that are reasonably certain to occur within the action area in the reasonably foreseeable future. SURTASS LFA sonar activities in the action area proposed from August 15, 2016, through August 14, 2017, are expected to continue at similar levels into the reasonably foreseeable future. Many of these activities will occur without any humpback whales being exposed to sound fields associated with LFA sonar. 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 and conference report, some of the primary anthropogenic threats to the survival and recovery of humpback whales have been whaling and ship strikes. The threat of whaling has been eliminated. The current abundance trend for humpback whales rangewide including the action area is not well understood.

As described in section 6.4.5 of this opinion and conference report, assuming that the U.S. Navy conducts the maximum extent of routine training, testing, and military operational activities in a year, we estimated there will be approximately 184 instances of humpback whale behavioral harassment from SURTASS LFA sonar. The whales will be affected over the course of 17 proposed missions and will be from four stocks, the Western North Pacific, Central North Pacific, Arabian Sea, and Western Australia stocks. The most affected of the humpback stocks is predicted to be the Western North Pacific stock, which is composed of 1,328 individuals, and would experience 184 instances of behavioral harassment during 17 missions.

The evidence available suggests that humpback whales, like other baleen whales, exposed to SURTASS LFA sonar transmissions are not likely to be killed or experience injury, masking, stranding, resonance effects, or behavioral responses that might reduce the longevity or reproductive success of individuals that have been exposed. Instead, the best scientific and commercial data available suggests that SURTASS LFA sonar transmissions are likely to elicit short-term behavioral responses in humpback whales that are known to have no long-term, adverse consequences for the biology or ecology of the individual whales exposed to an LFA sonar signal.

We do not anticipate any take in the form of injury from PTS or other injuries such as GI tract or lung injury during routine training, testing, or military operations. We do not anticipate any mortality of humpback whales from SURTASS LFA sonar stressors. The estimates of exposures to LFA sonar signals that would result in behavioral responses annually 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.

Considering information presented in this opinion and conference report, we consider humpback whales to be able to hear and respond to low-frequency sonar similar to blue whales. Most of the humpback whales that are exposed to acoustic stressors would be exposed periodically or episodically over certain months or seasons from SURTASS LFA in the action area. Given the nature of LFA sonar routine testing, training, and military operations as described above, these periodic or episodic exposure and behavioral response scenarios most often allow sufficient time to return to baseline conditions and resumption of normal behavioral activities such as feeding, breeding, and migrating. As described previously in this opinion and conference report, the available scientific information does not provide evidence that exposure to acoustic stressors from LFA sonar will impact the fitness of any individuals of this species. Therefore exposure to acoustic stressors will not have population or species level impacts.

The general increase in the number of humpback whales range-wide suggests that the stress regime these whales are exposed to including activities in the action area have not prevented these whales from increasing their numbers. Humpback whales have been exposed to U.S. Navy routine training, testing, and military operations in the action area, including vessel traffic, aircraft traffic, active sonar, and underwater detonations, for more than a generation.

The 1991 humpback whale recovery plan does not outline specific downlisting and delisting criteria. The recovery plan does list several threats known or suspected of impacting humpback whale recovery including subsistence hunting, commercial fishing stressors, habitat degradation, loss of prey species, ship collision, and acoustic disturbance. Of these, ship collision and acoustic disturbance are relevant to SURTASS LFA sonar activities. As described previously, anthropogenic noise associated with SURTASS LFA sonar activities will not impact the fitness of any individuals of this species.

Based on the evidence available, including the *Environmental Baseline* and *Cumulative Effects*, stressors resulting from SURTASS LFA sonar routine training, testing, and military operations the U.S. Navy will conduct in the action area from August 15, 2016 through August 14, 2017 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, directly or indirectly, to appreciably reduce the likelihood of the survival of humpback whales in the wild by reducing the reproduction, numbers, or distribution of that species. We also conclude that effects from ongoing U.S. Navy SURTASS LFA sonar routine training, testing, and military operations continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of humpback whales in the wild by reducing the reproduction, numbers, or distribution of that species. SURTASS LFA sonar 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 is not 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) and therefore, we do not expect acoustic stressors to result in substantial changes in reproduction, numbers, or distribution of these populations.. By the same logic, we do not anticipate any reductions in survival rate or trajectory of recovery of the species as listed pursuant to the ESA that could be readily perceived or estimated.

Due to a lack of fitness consequences to individuals and the populations they represent, we also do not anticipate any reductions in survival rate or trajectory of recovery of the western North Pacific gray whale population as currently proposed.

7.5 North Pacific Right Whale

In determining whether U.S. Navy SURTASS LFA sonar routine training, testing and military operations in the action area are likely to jeopardize the survival and recovery of North Pacific right 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. SURTASS LFA sonar activities in the action area proposed from August 15, 2016, through August 14, 2017, are expected to continue at similar levels into the reasonably foreseeable future. Many of these activities will occur without any North Pacific right whales being exposed to sound fields associated with LFA sonar. 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 and conference report, some of the primary anthropogenic threats to the survival and recovery of North Pacific right whales have been whaling and ship strikes. The threat of whaling has been eliminated. The current abundance trend for North Pacific right whales rangewide including the action area is not well understood.

As described in section 6.4.5 of this opinion and conference report, assuming the U.S. Navy conducts the maximum extent of routine training, testing, and military operational activities in a year, we estimated there will be approximately seven instances of North Pacific right whale behavioral harassment from SURTASS LFA sonar. The western North Pacific stock of right whales is estimated to include 922 animals. The seven instances of behavioral harassment would result in 0.22 percent of the western North Pacific stock being affected by SURTASS LFA sonar transmissions during eight missions.

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, testing, or operations. We do not anticipate any mortality of North Pacific right whales from SURTASS LFA sonar stressors. The estimates of exposures to LFA sonar signals that would result in behavioral responses annually would not be expected to appreciably reduce the likelihood of the survival of North Pacific right whales in the wild by reducing the reproduction, numbers, or distribution of that species for the reasons discussed herein.

Considering information presented in this opinion and conference report, we consider North Pacific right whales to be able to hear and respond to low-frequency sonar similar to blue whales. Most of the North Pacific right whales that are exposed to acoustic stressors would be exposed periodically or episodically over certain months or seasons from military activities in the SURTASS LFA sonar action area. Given the nature of LFA sonar routine training, testing, and military operations as described above, these periodic or episodic exposure and behavioral response scenarios most often allow sufficient time to return to baseline conditions and resumption of normal behavioral activities such as feeding, breeding, and migrating. As described previously in this opinion and conference report, the available scientific information does not provide evidence that exposure to acoustic stressors from LFA sonar will impact the fitness of any individuals of this species. Therefore exposure to acoustic stressors will not have population or species level impacts.

The 2013 North Pacific right whale recovery plan defines two recovery populations in the North Pacific Ocean (the western and eastern) and sets criteria for the downlisting and delisting of this species. The recovery plan lists several stressors potentially affecting the status of North Pacific right whales that are relevant to SURTASS LFA sonar activities including: vessel disturbance and military operations (including sonar). As discussed previously, anthropogenic noise associated with SURTASS LFA sonar activities will not impact the fitness of any individuals of this species. Downlisting criteria for North Pacific right whales includes the maintenance of at least 250 mature females and 250 mature males in each recovery population. To qualify for downlisting, each recovery population must also have no more than a one percent chance of extinction in 100 years. To qualify for delisting, each recovery population must also have no more than a ten percent chance of becoming endangered in 25 years. To our knowledge a population viability analysis has not been conducted on North Pacific right whale recovery populations.

Based on the evidence available, including the *Environmental Baseline* and *Cumulative Effects*, stressors resulting from SURTASS LFA sonar routine training, testing, and military operations the U.S. Navy will conduct in the action area from August 15, 2016 through August 14, 2017 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, directly or indirectly, to appreciably reduce the likelihood of the survival of North Pacific right whales in the wild by reducing the reproduction, numbers, or distribution of that species. We also conclude that effects from ongoing U.S. Navy SURTASS LFA sonar routine training, testing, and military operations continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of North Pacific right whales in the wild by reducing the reproduction, numbers, or distribution of that species. SURTASS LFA sonar stressors will not affect the population dynamics, behavioral ecology, and social dynamics of individual right whales in ways or to a degree that would reduce their fitness. An action that is not likely to reduce the fitness of individual whales is not 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) and therefore, we do not expect acoustic stressors to result in substantial changes in reproduction, numbers, or distribution of these populations.. By the same logic, we do not anticipate any reductions in survival rate or trajectory of recovery of the species as listed pursuant to the ESA that could be readily perceived or estimated.

7.6 Sei Whale

In determining whether U.S. Navy SURTASS LFA sonar routine training, testing and military operations in the 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. SURTASS LFA sonar activities in the action area proposed from August 15, 2016, through August 14, 2017, are expected to continue at similar levels into the reasonably foreseeable future. Many of these activities will occur without any sei whales being exposed to sound fields associated with LFA sonar. 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 and conference report, some of the primary anthropogenic threats to the survival and recovery of sei whales have been whaling and ship strikes. The threat of whaling has been eliminated. The current abundance trend for sei whales rangewide including the action area is not well understood.

As described in section 6.4.5 of this opinion and conference report, assuming the U.S. Navy conducts the maximum extent of routine training, testing, and military operations in a year, we estimated there will be approximately 136 instances of sei whale behavioral harassment from SURTASS LFA sonar during ten missions. The instances of behavioral harassment would result in 1.81 percent of the North Pacific stock being affected by SURTASS LFA sonar transmissions during six missions and 2.20 percent of the Hawaii stock being affected during nine missions.

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, testing, or operations. We do not anticipate any mortality of sei whales from SURTASS LFA sonar stressors. The estimates of exposures to LFA sonar signals that would result in behavioral responses annually 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.

Considering information presented in this opinion and conference report, we consider sei whales to be able to hear and respond to low-frequency sonar similar to blue whales. Most of the sei whales that are exposed to acoustic stressors would be exposed periodically or episodically over certain months or seasons from military activities in the SURTASS LFA sonar action area. Given the nature of SURTASS LFA routine sonar training, testing, and military operations as described above, these periodic or episodic exposure and behavioral response scenarios most often allow sufficient time to return to baseline conditions and resumption of normal behavioral activities such as feeding and breeding. As described previously in this opinion and conference report, the available scientific information does not provide evidence that exposure to acoustic stressors from LFA sonar 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 SURTASS LFA sonar activities. As described previously, anthropogenic noise associated with SURTASS LFA sonar 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 one percent chance of extinction in 100 years. To qualify for delisting, each recovery population must also have no more than a ten 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 SURTASS LFA sonar routine training, testing, and military operational activities the U.S. Navy will conduct in the action area from August 15, 2016, through August 14, 2017, 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, directly or indirectly, to appreciably reduce the likelihood of the survival of sei whales in the wild by reducing the reproduction, numbers, or distribution of that species. We also conclude that effects from ongoing U.S. Navy SURTASS LFA sonar routine training, testing, and military operational 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. SURTASS LFA sonar 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 is not 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) and therefore, we do not expect acoustic stressors to result in substantial changes in reproduction, numbers, or distribution of these populations.. By the same logic, we do not anticipate any reductions in survival rate or trajectory of recovery of the species as listed pursuant to the ESA that could be readily perceived or estimated.

7.7 Sperm Whale

In determining whether U.S. Navy SURTASS LFA sonar routine training, testing and military operational activities in the 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. SURTASS LFA sonar activities in the action area proposed from August 15, 2016, through August 14, 2017, are expected to continue at similar levels into the reasonably foreseeable future. Many of these activities will occur without any sperm whales being exposed to sound fields associated with LFA sonar. 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 and conference report, some of the primary anthropogenic threats to the survival and recovery of sperm whales have been whaling and ship strikes. The threat of whaling has been eliminated. The current abundance trend for fin whales rangewide including the action area is not well understood.

As described in section 6.4.5 of this opinion and conference report, assuming that the U.S. Navy conducts the maximum extent of routine training, testing, and military operational activities in a year, we estimated there will be approximately 392 instances of sperm whale behavioral harassment from SURTASS LFA sonar during 20 nominal missions. Based on the results of the U.S. Navy's modeling, the North Pacific stock that would experience the highest number of instances of behavioral harassment with 329 (affecting 0.31 percent of the stock). The Hawaiian stock of sperm whales, consisting of approximately 3,354 individuals will also be affected, with 32 instances of behavioral harassment (affecting 0.92 percent of the stock). The North Indian stock of sperm whales, consisting of approximately 24,446 individuals, and the South Indian stock of sperm whales, consisting of approximately 24,446 individuals, will be affected with 25 and 6 instances of behavioral harassment, respectively (affecting 0.10 and 0.02 percent of the stocks, respectively).

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 routine training, testing, or military operations. We do not anticipate any mortality of sperm whales from SURTASS LFA sonar stressors. The estimates of exposures to LFA sonar signals that would result in behavioral responses 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.

Most of the sperm whales that are exposed to acoustic stressors would be exposed periodically or episodically over certain months or seasons from military activities in the SURTASS LFA sonar action area. Given the nature of LFA sonar routine testing, training, and military operations as described above, these periodic or episodic exposure and behavioral response scenarios most often allow sufficient time to return to baseline conditions and resumption of normal behavioral activities such as feeding and breeding. As described previously in this opinion and conference report, the available scientific information does not provide evidence that exposure to acoustic stressors from LFA sonar 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 SURTASS LFA sonar activities. As discussed previously, anthropogenic noise associated with SURTASS LFA sonar 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 one percent chance of extinction in 100 years. To qualify for delisting, each recovery population must also have no more than a ten percent chance of becoming endangered in 20 years. To our knowledge a population viability analysis has not been conducted on sperm whale recovery populations.

Based on the evidence available, including the *Environmental Baseline* and *Cumulative Effects*, stressors resulting from SURTASS LFA sonar routine training, testing, and military operational activities the U.S. Navy will conduct in the action area from August 15, 2016 through August 14,2017 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, directly or indirectly, to appreciably reduce the likelihood of the survival of sperm whales in the wild by reducing the reproduction, numbers, or distribution of that species. We also conclude that effects from ongoing U.S. Navy SURTASS LFA sonar routine training, testing, and military operational 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. SURTASS LFA sonar 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 is not 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) and therefore, we do not expect acoustic stressors to result in substantial changes in reproduction, numbers, or distribution of these populations.. By the same logic, we do not anticipate any reductions in survival rate or trajectory of recovery of the species as listed pursuant to the ESA that could be readily perceived or estimated.

7.8 Main Hawaiian Islands Insular DPS of False Killer Whale

In determining whether U.S. Navy SURTASS LFA sonar routine training, testing and military operational activities in the action area are likely to jeopardize the survival and recovery of MHI insular DPS of false killer whales, we assessed effects of the action against the aggregate effects of everything in the *Environmental Baseline* that has led to the current *Status of Listed Resources* and, those effects of future non-Federal activities that are reasonably certain to occur within the action area in the reasonably foreseeable future. SURTASS LFA sonar activities in the action area proposed from August 15, 2016, through August 14, 2017, are expected to continue at similar levels into the reasonably foreseeable future. Many of these activities will occur without any MHI insular false killer whales being exposed to sound fields associated with LFA sonar. 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 and conference report, several threats have been identified that may have or continue to lead to the decline of Main Hawaiian Islands insular false killer whales. Also, reduced genetic diversity may be a natural, but partially anthropogenically induced factor leading to MHI insular false killer whale decline (Wearmouth and Sims 2008). These include incidental interactions with commercial and recreational fisheries and aquaculture facilities, prey availability, vessel traffic, anthropogenic noise, small populations effects, disease and predation, parasitism, environmental contaminants, harmful algal blooms, and ocean acidification and climate change (Oleson et al. 2010). The current abundance trend for MHI insular false killer whales in the action area is not well understood. Some data indicate a slight increase.

As described in section 6.4.5 of this opinion and conference report, assuming that the U.S. Navy conducts the maximum extent of routine training, testing, and military operational activities in a year, we estimated there will be approximately three instances of MHI insular false killer whale behavioral harassment from SURTASS LFA sonar during two missions. This would occur during one mission in the Hawaii-North (1.27 percent of the stock affected) and one mission in the Hawaii-South (0.05 percent of the stock affected) mission areas. Abundance for MHI insular false killer whales is estimated at 151 individuals. In total, approximately 1.32 percent of the of the population of the MHI insular DPS of false killer whales would be affected by SURTASS LFA sonar transmissions.

The evidence available suggests, however, that MHI insular false killer whales, like other toothed whales, are not very sensitive to low-frequency sounds. Despite the limited number of studies, the available evidence suggests that the risk of injury, masking, stranding, resonance effects, or behavioral effects in these whales is very low. The best scientific and commercial data available suggests that SURTASS LFA sonar transmissions are likely to elicit short-term behavioral effects on MHI insular false killer whales that are known to have no long-term, adverse consequences for the biology or ecology of the individual whales exposed to the LFA sonar signal.

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 routine training, testing, or military operational activities. We do not anticipate any mortality of MHI insular DPS of false killer whales from SURTASS LFA sonar stressors. The estimates of exposures to LFA sonar signals that would result in behavioral responses annually would not be expected to appreciably reduce the likelihood of the survival of MHI insular false killer whales in the wild by reducing the reproduction, numbers, or distribution of that species for the reasons discussed herein.

The MHI insular DPS of false killer whale does not have a recovery plan; therefore, specific downlisting and delisting criteria are not established. We concluded no mortality of this listed entity would occur and that effects from acoustic stressors would be temporary and not impact the fitness of individuals or the population. In the absence of fitness consequences on individuals or the population to which those individuals belong, we do not expect an appreciable reduction in the ability of this entity to recover.

Based on the evidence available, including the *Environmental Baseline* and *Cumulative Effects*, stressors resulting from U.S. Navy SURTASS LFA sonar routine annual training, testing, and military operational activities proposed August 15, 2016, through August 14, 2017, 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, directly or indirectly, to appreciably reduce the likelihood of the survival of MHI insular false killer whales in the wild by reducing the reproduction, numbers, or distribution of that species. We also conclude that effects from ongoing SURTASS LFA sonar activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of MHI insular false killer whales in the wild by reducing the reproduction, numbers, or distribution of that species. SURTASS LFA sonar stressors will not affect the population dynamics, behavioral ecology, and social dynamics of individual false killer whales in ways or to a degree that would reduce their fitness. An action that is not likely to reduce the fitness of individual whales is not 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) and therefore, we do not expect acoustic stressors to result in substantial changes in reproduction, numbers, or distribution of these populations. By the same logic, we do not anticipate any reductions in survival rate or trajectory of recovery of the species as listed pursuant to the ESA that could be readily perceived or estimated.

7.9 Hawaiian Monk Seal

In determining whether U.S. Navy SURTASS LFA sonar routine training, testing and military operational activities in the action area are likely to jeopardize the survival and recovery of the Hawaiian monk seal, 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. SURTASS LFA sonar activities in the action area proposed from August 15, 2016, through August 14, 2017, are expected to continue at similar levels into the reasonably foreseeable future. Many of these activities will occur without any Hawaiian monk seals being exposed to sound fields associated with LFA sonar. 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 and conference report, several threats have been identified that may have or continue to lead to the decline of Hawaiian monk seals. Monk seals are threatened by natural predation, disease outbreaks, biotoxins, and agonistic behavior by male monk seals (NMFS 2011c). Monk seals, particularly pups, are also subjected to extensive predation by sharks, which appear to be a significant problem for the monk seals occupying French Frigate Shoals in the Northwest Hawaiian Islands (Antonelis et al. 2006). One of the most substantial threats to Hawaiian monk seals results from dramatic declines in the survival of juveniles and appears to be related to significantly reduced body sizes in pup and juvenile seals. Several human activities are known to threaten Hawaiian monk seals: commercial and subsistence hunting, intentional harassment, competition with commercial fisheries, entanglement in fishing gear, habitat destruction on breeding beaches, pollution, and unintentional human disturbance (Kenyon 1981; Reeves et al. 1992; Riedman 1990). The revised recovery plan for Hawaiian monk seals identifies food limitation, entanglements, and shark predation as crucial threats to the continued existence of this species (NMFS 2007d).

As described in section 6.4.5 of this opinion and conference report, assuming that the U.S. Navy conducts the maximum extent of routine training, testing, and military operational activities in a year, we estimated there will be approximately eight instances of Hawaiian monk seal behavioral harassment from SURTASS LFA sonar during three missions. This would occur in the Hawaii North, Hawaii South, and Offshore North Pacific (25 to 40^{o} North) mission areas. Abundance for Hawaiian monk seals is estimated at 1,153 individuals. Approximately 0.49 percent of the of the Hawaiian monk seal population would be affected by SURTASS LFA sonar transmissions during the three missions.

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 routine training, testing, or military operations. We do not anticipate any mortality of Hawaiian monk seals from SURTASS LFA sonar stressors. The estimates of exposures to LFA sonar signals that would result in behavioral responses annually would not be expected to appreciably reduce the likelihood of the survival of Hawaiian monk seals in the wild by reducing the reproduction, numbers, or distribution of that species for the reasons discussed herein.

The 2007 Hawaiian monk seal recovery plan defines seven recovery populations, the French Frigate Shoals, Lisianski Island, Midway Atoll, Laysan Island, Pearl and Hermes Reef, Kure Atoll, and Main Hawaiian Islands, and sets criteria for the downlisting and delisting of this species. Both downlisting and delisting requirements include abatement of threats associated with food limitation, debris entanglement, predation, fisheries, erosion, and disease. None of these stressors have been determined to be relevant to SURTASS activities with regards to this species. Downlisting criteria for Hawaiian monk seals includes the maintenance of 2,900 individuals in the Northwestern Hawaiian Islands (NWHI), at least five of the six NWHI sub-populations with at least 100 individuals and the MHI population above 500 individuals, and a population growth rate of zero or higher. To qualify for delisting, the downlisting criteria must be met for 20 consecutive years. As described previously, we conclude the temporary effects from acoustic stressors associated with U.S. Navy SURTASS LFA activities will not impact the fitness of any individuals of this species or the populations to which they belong.

Based on the evidence available, including the *Environmental Baseline* and *Cumulative Effects*, stressors resulting from U.S. Navy SURTASS LFA sonar annual routine training, testing, and military operational activities proposed August 15, 2016, through August 14, 2017, 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, directly or indirectly, to appreciably reduce the likelihood of the survival of Hawaiian monk seals in the wild by reducing the reproduction, numbers, or distribution of that species. We also conclude that effects from ongoing SURTASS LFA sonar activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of Hawaiian monk seals in the wild by reducing the reproduction, numbers, or distribution of that species. SURTASS LFA sonar stressors will not affect the population dynamics, behavioral ecology, and social dynamics of individual Hawaiian monk seals in ways or to a degree that would reduce their fitness. An action that is not likely to reduce the fitness of individual seals is not likely to reduce the viability of the populations those individual seals compose (that is, we would not expect reductions in the reproduction, numbers, or distribution of those populations) and therefore, we do not expect acoustic stressors to result in substantial changes in reproduction, numbers, or distribution of these populations. By the same logic, we do not anticipate any reductions in survival rate or trajectory of recovery of the species as listed pursuant to the ESA that could be readily perceived or estimated.

7.10 Southern DPS of Spotted Seal

In determining whether U.S. Navy SURTASS LFA sonar routine training, testing and military operational activities in the action area are likely to jeopardize the survival and recovery of the Southern DPS of the spotted seal, 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. SURTASS LFA sonar activities in the action area proposed from August 15, 2016, through August 14, 2017, are expected to continue at similar levels into the reasonably foreseeable future. Many of these activities will occur without any spotted seals being exposed to sound fields associated with LFA sonar. 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 and conference report, several threats have been identified that may have or continue to lead to the decline of spotted seals. Spotted seals have long been a target of commercial and subsistence hunting, but threats from climate change and environmental contaminants are more recent developments. Although significant direct effects from climate change are not expected, indirect effects can be important (Boily 1995; Grebmeier et al. 2006; Harding et al. 2005). These include changes in prey abundance or distribution, predation, and disease (Boveng et al. 2009a; Comeau et al. 2009; Grebmeier et al. 2006). Additionally, spotted seals are still hunted for meat, fur, oil, and animal feed by coastal inhabitants of the northern Pacific Rim (Fedoseev 1984; Krylov et al. 1964).

As described in section 6.4.5 of this opinion and conference report, assuming that the U.S. Navy conducts the maximum extent of routine training, testing, and military operational activities in a year, we estimated there will be approximately two instances of spotted seal behavioral harassment from SURTASS LFA sonar during two missions. This would occur during the missions in the Sea of Japan and East China Sea mission areas. Abundance for the Southern DPS of spotted seals is estimated at 1,000 individuals. Approximately 0.02 percent of the Southern DPS of spotted seals would be affected by SURTASS LFA sonar transmissions during the two missions.

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, testing, or operations. We do not anticipate any mortality of spotted seals from SURTASS LFA sonar stressors. The estimates of exposures to LFA sonar signals that would result in behavioral responses annually would not be expected to appreciably reduce the likelihood of the survival of spotted seals in the wild by reducing the reproduction, numbers, or distribution of that species for the reasons discussed herein.

The Southern DPS of spotted seals does not have a recovery plan; therefore, specific downlisting and delisting criteria are not established. We concluded no mortality of this listed entity would occur and that effects from acoustic stressors would be temporary and not impact the fitness of individuals or the population. In the absence of fitness consequences on individuals or the population to which those individuals belong, we do not expect an appreciable reduction in the ability of this entity to recover.

Based on the evidence available, including the *Environmental Baseline* and *Cumulative Effects*, stressors resulting from U.S. Navy SURTASS LFA sonar annual routine training, testing, and military operational activities proposed August 15, 2016, through August 14, 2017, 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, directly or indirectly, to appreciably reduce the likelihood of the survival of the Southern DPS of spotted seals in the wild by reducing the reproduction, numbers, or distribution of that species. We also conclude that effects from ongoing SURTASS LFA sonar activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of the Southern DPS of spotted seals in the wild by reducing the reproduction, numbers, or distribution of that species. SURTASS LFA sonar stressors will not affect the population dynamics, behavioral ecology, and social dynamics of individual spotted seals in ways or to a degree that would reduce their fitness. An action that is not likely to reduce the fitness of individual seals is not likely to reduce the viability of the populations those individual seals compose (that is, we would not expect reductions in the reproduction, numbers, or distribution of those populations) and therefore, we do not expect acoustic stressors to result in substantial changes in reproduction, numbers, or distribution of these populations. By the same logic, we do not anticipate any reductions in survival rate or trajectory of recovery of the species as listed pursuant to the ESA that could be readily perceived or estimated.

7.11 Western DPS of Steller Sea Lion

In determining whether U.S. Navy SURTASS LFA sonar routine training, testing and military operational activities in the action area are likely to jeopardize the survival and recovery of the Western DPS of the Steller sea lion, 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. SURTASS LFA sonar activities in the action area proposed from August 15, 2016, through August 14, 2017, are expected to continue at similar levels into the reasonably foreseeable future. Many of these activities will occur without any Steller sea lions being exposed to sound fields associated with LFA sonar. 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 and conference report, several threats have been identified that may have or continue to lead to the decline of Steller sea lions including, but not limited to, harvest, competition for prey with fisheries, and contaminants.

As described in section 6.4.5 of this opinion and conference report, assuming that the U.S. Navy conducts the maximum extent of routine training, testing, and military operational activities in a year, we estimated there would be approximately one instance of Steller sea lion behavioral harassment from SURTASS LFA sonar during 0.5 missions. This would occur during the missions in the Northeast of Japan mission area. Abundance for the Western DPS of Steller sea lions is estimated at 82,516 individuals. Less than 0.01 percent of the Western DPS of Steller sea lions would be affected by SURTASS LFA sonar transmissions during the 0.5 missions.

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, testing, or operations. We do not anticipate any mortality of Steller sea lions from SURTASS LFA sonar stressors. The estimates of exposures to LFA sonar signals that would result in behavioral responses annually would not be expected to appreciably reduce the likelihood of the survival of Steller sea lions in the wild by reducing the reproduction, numbers, or distribution of that species for the reasons discussed herein.

Based on the evidence available, including the *Environmental Baseline* and *Cumulative Effects*, stressors resulting from U.S. Navy SURTASS LFA sonar annual routine training, testing, and military operational activities proposed August 15, 2016, through August 14, 2017, 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, directly or indirectly, to appreciably reduce the likelihood of the survival of the Western DPS of Steller sea lions in the wild by reducing the reproduction, numbers, or distribution of that species. SURTASS LFA sonar stressors would not affect the population dynamics, behavioral ecology, and social dynamics of individual Steller sea lions in ways or to a degree that would reduce their fitness. An action that is not likely to reduce the fitness of individual sea lions is not likely to reduce the viability of the populations those individual sea lions compose (that is, we would not expect reductions in the reproduction, numbers, or distribution of those populations) and therefore, we do not expect acoustic stressors to result in substantial changes in reproduction, numbers, or distribution of these populations. By the same logic, we do not anticipate any reductions in survival rate or trajectory of recovery of the species as listed pursuant to the ESA that could be readily perceived or estimated.

7.12 Sea Turtles

In determining whether U.S. Navy SURTASS LFA sonar routine training, testing and military operational activities in the action area are likely to jeopardize the survival and recovery of the ESA-listed 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. SURTASS LFA sonar activities in the action area proposed from August 15, 2016, through August 14, 2017, are expected to continue at similar levels into the reasonably foreseeable future. Many of these activities would occur without any sea turtles being exposed to sound fields associated with LFA sonar. 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 and conference report, some of the primary anthropogenic threats to ESA-listed sea turtles are fisheries interactions, climate change, and impacts to terrestrial nesting habitat. SURTASS LFA sonar (or other military sonars) has not been identified as a primary threat to ESA-listed sea turtles.

Due to the lack of more definitive data on sea turtle population distributions in the open ocean, we were not able to estimate the percentage of a sea turtle population (or number of individuals) that could be located in a SURTASS LFA sonar operations area at a potentially vulnerable depth, during an LFA sonar sound transmission. We determined that due to the monitoring and mitigation measures employed during SURTASS LFA sonar transmissions that will reduce the likelihood of sea turtles being in close proximity to the sound source, any exposure of sea turtles to LFA sonar signals is not expected to result in mortality, injury (including PTS), or TTS. We anticipate sea turtles exposed to LFA sonar could result in instances of behavioral harassment, but determined that we do not expect sea turtles would exhibit long term behavioral responses that could impact an individual’s fitness if they were exposed to SURTASS LFA sonar. Because we do not expect instances of behavioral harassment to result in fitness consequences to individual ESA-listed sea turtles, we do not expect population level effects from exposure to SURTASS LFA sonar.

Based on the evidence available, including the *Environmental Baseline* and *Cumulative Effects*, stressors resulting from U.S. Navy SURTASS LFA sonar annual routine training, testing, and military operational activities proposed August 15, 2016, through August 14, 2017, or cumulatively for the reasonably foreseeable future, would not be expected, directly or indirectly, to appreciably reduce the likelihood of the survival of ESA-listed sea turtles in the wild by reducing the reproduction, numbers, or distribution of the sea turtle species considered in this opinion. We also conclude that effects from ongoing SURTASS LFA sonar activities continuing into the reasonably foreseeable future would not be expected, directly or indirectly, to appreciably reduce the likelihood of recovery of ESA-listed sea turtles in the wild by reducing the reproduction, numbers, or distribution of the sea turtle species considered in this opinion. SURTASS LFA sonar stressors would not affect the population dynamics, behavioral ecology, and social dynamics of individual ESA-listed sea turtles in ways or to a degree that would reduce their fitness. An action that is not likely to reduce the fitness of individual sea turtles is not likely to reduce the viability of the populations those individual turtles compose (that is, we would not expect reductions in the reproduction, numbers, or distribution of those populations) and therefore, we do not expect acoustic stressors to result in substantial changes in reproduction, numbers, or distribution of these populations. By the same logic, we do not anticipate any reductions in survival rate or trajectory of recovery of these species as listed pursuant to the ESA that could be readily perceived or estimated.

8 CONCLUSION

During the consultation, we reviewed the current status of endangered blue, fin, western North Pacific gray, humpback, North Pacific right, sei, sperm, MHI insular DPS of false killer whales, Hawaiian monk seals, Western DPS of Steller sea lions, Central West Pacific DPS of green, hawksbill, leatherback, and North Pacific Ocean DPS and Southeast Indian Pacific Ocean DPS of loggerhead sea turtles; as well as the threatened Southern DPS of spotted seals, East Indian-West Pacific DPS, Central North Pacific DPS, and North Indian DPS of green, North Indian Ocean DPS of loggerhead, and olive ridley sea turtles. Additionally, we assessed the status of the proposed endangered Arabian Sea DPS and threatened western North Pacific DPS of humpback whales, and the proposed extension of Hawaiian monk seal critical habitat as part of our conference report. We also assessed the *Environmental Baseline* within the action area, the effects of the proposed action, any effects of interrelated and interdependent actions, and cumulative effects.

While the number of instances of behavioral harassment (incidental “take”) gets larger over time, the effect of each “take” on the survival or reproductive success of the animals themselves would not accumulate. As a result, we do not expect that instances of exposing whales or other ESA-listed species to SURTASS LFA sonar in a single year, or instances of exposing them to LFA sonar over the remaining period of the MMPA rule or into the reasonably foreseeable future, would result in effects that would be greater than what we would expect from a single exposure event. To the contrary, we did not expect the effects of that “take” to have any additive, interactive, or synergistic effect on the individual animals, the population(s) those individuals represent, or the species those population(s) comprise. With respect to threatened and endangered marine mammals, our conclusion that the aggregate number of exposures over the duration of the MMPA regulations or and into the reasonably foreseeable future is unlikely to result in accumulated adverse impacts is also supported by the negligible impact determination and response to comments contained in the MMPA rulemaking.

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”). For the reasons set forth above, and taking into consideration the best available scientific evidence documented throughout this opinion and conference reportreport, we conclude that the continuation of SURTASS LFA sonar routine training, testing, and military operational activities into the reasonable foreseeable future, at the levels described in the current five-year MMPA rule (and assuming no change in the *Status of ESA-listed Resources* or the *Environmental Baseline*), are unlikely to lead to any adverse, long-term additive or cumulative impacts on individuals or affected species, and that such long-term impacts are not reasonably certain to occur based on the information that is currently available. Furthermore, our analysis and conclusions in this opinion and conference report are based on modeled estimates of exposures and takes, assuming that the U.S. Navy conducts the maximum number of authorized missions, for the maximum number of authorized sonar transmit hours. Therefore, our assumption that the U.S. Navy's routine training, testing, and military operations using SURTASS LFA sonar will continue into the reasonably foreseeable future does not alter our conclusion that the U.S. Navy's activities are unlikely to jeopardize the continued existence of any ESA-listed species or destroy or adversely modify critical habitat that has been designated for such species.

Therefore, it is NMFS' opinion that SURTASS LFA sonar routine training, testing, and military operations conducted from August 15, 2016, to August 14, 2017, or into the reasonably foreseeable future, assuming that the type, amount and extent of training, testing, and military operations do not exceed levels assessed in this opinion and conference report and/or the status of the species affected by these actions does not change significantly from that assessed in this opinion and conference report are likely to adversely affect, but are not likely to jeopardize the continued existence of, blue, fin, western North Pacific gray, humpback, North Pacific right, sei, sperm, MHI insular DPS of false killer whales, Hawaiian monk seals, Southern DPS of spotted seals, or Western DPS of Steller sea lions. We find that these same proposed actions are not likely to adversely affect, nor jeopardize the continued existence of, Indo-West Pacific DPS of scalloped hammerhead sharks, Chinese sturgeon, Sakhalin sturgeon, Chinook salmon, or sea turtles. The same proposed actions will not destroy or adversely modify Western DPS of Steller sea lion, Hawaiian monk seal, and North Pacific right whale designated critical habitat.

Additionally, if the proposed ESA-listing is finalized as proposed and assessed in this opinion and conference report, we conclude that SURTASS LFA sonar routine training, testing, and military operations conducted from August 15, 2016 to August 14, 2017, or into the reasonably foreseeable future, assuming that the type, amount and extent of training, testing, and military operations do not exceed levels assessed in this opinion and conference report and/or the status of the species affected by these actions does not change significantly from that assessed in this conference report, would likely adversely affect, but not jeopardize the continued existence of, the proposed endangered Arabian Sea DPS and threatened western North Pacific DPS of humpback whales. If the proposed ESA-listing is finalized as proposed and assessed in this opinion and conference report, we conclude that these same proposed actions would not destroy or adversely modify, the proposed extension of Hawaiian monk seal designated critical habitat.

This opinion and conference report also concludes that the NMFS' issuance of the LOAs pursuant to the MMPA five-year rule for the U.S. Navy to take marine mammals for a period beginning in August 15, 2016, and ending in August 14, 2017, incidental to SURTASS LFA sonar routine training, testing, and military operations onboard four vessels are likely to adversely affect but are not likely to jeopardize the continued existence of these threatened and endangered species under NMFS' jurisdiction and are not likely to result in the destruction or adverse modification of critical habitat that has been designated for endangered or threatened species in the action area.

9 INCIDENTAL TAKE STATEMENT

Section 9 of the ESA and Federal regulations pursuant to section 4(d) of the ESA prohibit the take of endangered and threatened species, respectively, without a special exemption. “Take” is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct. Harm is further defined by regulation to include significant habitat modification or degradation that results in death or injury to ESA-listed species by significantly impairing essential behavioral patterns, including breeding, feeding, or sheltering. Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. For this consultation, we interpret “harass” to mean an intentional or negligent action that has the potential to injure an animal or disrupt its normal behaviors to a point where such behaviors are abandoned or significantly altered.[12] 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.

ESA section 7(b)(4) states that take of ESA-listed marine mammals must be authorized under MMPA section 101(a)(5) before the Secretary can issue an incidental take statement for ESA-listed marine mammals. NMFS' implementing regulations for section 101 (a)(5)(A) specify that an LOA is required to conduct activities pursuant to any regulations for a specific activity that will "take" marine mammals. NMFS has authorized the incidental take of marine mammals under LOAs pursuant to 50 CFR § 218 Subpart X.

9.1 Amount or Extent of Take

The analysis contained in this opinion and conference report concluded that individual blue whales, fin whales, western North Pacific gray whales, humpback whales, North Pacific right whales, sei whales, sperm whales, MHI insular DPS of false killer whales, Hawaiian monk seals, Southern DPS of spotted seals, and Western DPS of Steller sea lions will be exposed to and adversely affected by SURTASS LFA sonar transmissions in the western and central North Pacific Ocean and Indian Ocean. As part of our conference report, we concluded that if the proposed rulemaking becomes effective and if the proposed listing is finalized as proposed and assessed in this opinion and conference report, individuals from the proposed endangered

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[12] NMFS has not adopted a regulatory definition of harassment under the ESA. The World English Dictionary defines harass as “to trouble, torment, or confuse by continual persistent attacks, questions, etc.” The U.S. Fish and Wildlife Service defines “harass” in its regulations as “an intentional or negligent act or omission which creates the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavioral patterns which include, but are not limited to, breeding, feeding, or sheltering (50 CFR 17.3). The interpretation we adopt in this consultation is consistent with our understanding of the dictionary definition of harass and is consistent with the Service’s interpretation of the term.

Arabian Sea DPS and threatened western North Pacific DPS of humpback whales would similarly be exposed to and adversely affected by SURTASS LFA sonar transmissions in the western and central North Pacific Ocean and Indian Ocean. Any animals that would be exposed to LFA sonar transmissions would occur in portions of the North Pacific Ocean and Indian Ocean:

- East of Japan; north Philippine Sea; west Philippine Sea; offshore Guam; the Sea of Japan; East China Sea; South China Sea; offshore Japan (25 to 40° North and 10 to 25° North), and northeast of Japan mission areas.
- The North and South Hawaii mission areas.
- Arabian Sea and Northwest of Australia mission areas.

Any threatened or endangered species that are exposed to LFA sonar transmissions may elicit behavioral responses that might be considered “harassment.” NMFS does not expect any threatened or endangered species to be injured or killed as a result of exposure to LFA sonar transmissions.

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 SURTASS LFA sonar transmissions was difficult to estimate because we have little empirical information on (a) the actual number of ESA-listed species that are likely to occur in the action area, (b) the actual number of individuals of those species that are likely to be exposed to LFA sonar transmissions, (c) the circumstances associated with any exposure, and (d) the range of responses we would expect different individuals of the different species to exhibit upon exposure.

Because this information was not available, this opinion and conference report relied on the U.S. Navy’s computer simulations to estimate the “number” of certain marine mammals that might be harassed during the employment of SURTASS LFA sonar; the results of these simulations appear in Section 6.4.5.

For the purposes of this opinion, conference report, and incidental take statement, we assumed that any non-zero value in Table 41. indicates that an individual whale has a probability of being exposed to received levels that might be expected to result in behavioral responses characteristic of an animal that has been harassed. That assumption results in the estimates shown in Table 41. below.

Table 41. The number of instances of behavioral harassment (incidental take) as a result of exposure to U.S. Navy SURTASS LFA sonar by mission area number.

Species	Estimated Annual Take by Species and Mission Area															Total Annual Take Estimates	
	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	Behavioral Harassment	Harm (Injury, PTS)
Number of Missions	***1.5***	***3***	***3***	***3***	***2***	***1***	***1***	***1***	***1***	***1***	***1***	**0.5**	**1**	**0.5**	**0.5**	**-**	**-**
Blue Whale	2	2	3	2	-	-	-	1	1	1	1	1	1	1	-	**16**	**0**
Fin Whale	3	11	11	2	31	-	4	1	1	1	1	2	1	13	21	**107**	**0**
Western North Pacific Gray Whale	-	-	-	-	1	1	1	-	-	-	-	-	-	-	1	**4**	**0**
Humpback Whale[1]	6	45	67	11	-	-	6	2	9	13	14	1	-	1	9	**184**	**0**
North Pacific Right Whale	1	2	-	-	0	1	1	-	-	-	-	-	-	-	2	**7**	**0**
Sei Whale	17	-	-	6	-	-	-	7	69	3	2	-	-	-	32	**136**	**0**
Sperm Whale	20	48	46	36	31	15	11	26	24	17	15	13	12	6	72	**392**	**0**
False Killer Whale – MHI Insular DPS	-	-	-	-	-	-	-	-	-	2	1	-	-	-	-	**3**	**0**
Hawaiian Monk Seal	-	-	-	-	-	-	-	4	-	2	2	-	-	-	-	**8**	**0**

Spotted Seal – Southern DPS	-	-	-	-	1	1	-	-	-	-	-	-	-	-	-	**2**	**0**
Steller Sea Lion – Western DPS	-	-	-	-	-	-	-	-	-	-	-	-	-	-	1	**1**	**0**
Sea Turtles	**	**	**	**	**	**	**	**	**	**	**	**	**	**	**	**	**0**

[1] During this consultation, NMFS proposed to divide the globally listed endangered species (humpback whale) into 14 DPSs and to remove the species-level listing, and in its place list two DPSs as endangered and two DPSs as threatened. If the proposed rulemaking becomes effective and if the proposed listings are finalized as proposed and assessed in this opinion and conference report, incidental take estimates for the range-wide species will become null and void and the estimated incidental take for the Arabian Sea DPS and Western North Pacific DPS will take effect.

*Behavioral Harrassment. We do not expect any instances of Temporary Threshold Shift (TTS) or PTS in sea turtles due to mitigation.

**Unspecified Number. While the potential for behavioral harrassment of sea turtles exists, it is very difficult to estimate the number and species composition of turtles that could be “taken.” Take will be exceeded if activity levels as proposed are exceeded or if the monitoring program detects any turtle species that have exposed to received levels greater than 180 dB, or if during or after LFA sonar operations it is determined that an animal was exposed to sound levels of 180 dB or higher, or if a vessel strike occurs.

We did not conduct computer simulations for sea turtles because the data necessary to develop computer models were not available. Due to the seasonal and life stage changes in habitat occupation, sea turtle populations are particularly difficult to census. Abundance estimates are based on the most current information available regarding counts of the most accessible member of the population, nesting females, which does not account for the abundance of male sea turtles. The lack of detailed population data and scarcity of density data for sea turtles do not allow for density estimates to be derived for populations in the North Pacific Ocean; therefore, we could not assign numerical limits for take estimates. Rather than specifying an amount of take for sea turtles, this incidental take statement specifies an extent of take as follows:

Adult and sub-adult sea turtles may be taken, in the form of harassment, in areas outside the LFA mitigation zone and the additional buffer zone required by the LOAs. (Because they tend to remain at or within a few meters of the ocean surface, we do not expect hatchling or juvenile sea turtles to be exposed to LFA sonar transmissions.) Because of turtle size and the density of the shell density, we assume that the U.S. Navy's monitoring programs, particularly the HF/M3 sonar system, would detect these larger sea turtles if they are in the mitigation zone.

Take of these species will have been exceeded if the monitoring program detects any individuals of these species that have been harmed, injured, or killed as result of exposure to LFA sonar transmissions (from which NMFS might infer that they had been exposed to received levels greater than 180 dB), or if during or after LFA sonar operations it is determined that an animal was exposed to sound levels of 180 dB or higher (i.e., it was not detected until after it was inside the 180 dB isopleth), or if a vessel strike occurs. Temporary threshold shift or PTS would not be expected to occur in sea turtles due to the mitigation.

Activity Levels Indicator of Take for Sea Turtles

Detection of behavioral responses of juvenile or adult sea turtles in coastal waters or at-sea during U.S. Navy SURTASS LFA training, testing, or operational activities would be extremely difficult. Most forms of behavioral responses would not be detected. Also, monitoring techniques to calculate actual take of including detection and collection of individuals and assessment of injuries or death is not feasible for sea turtles at the scale of SURTASS LFA sonar activities. Therefore, we must rely on predicted take associated with levels of activities and any opportunistic observations of potential behavioral responses or injured or dead juvenile or adult sea turtles during activities as measurements of take and a trigger for reinitiation of consultation. In the absence of observations of unanticipated levels of behavioral responses, injury or mortality, exceedance of an activity level will require the U.S. Navy to reinitiate consultation. Exceedances at the activity level or in other planned training events must be reported to NMFS prior to carrying out or immediately following, if reporting would interrupt U.S. Navy training activities.

9.2 Effects of the Take

In this opinion and conference report , NMFS determined that the amount or extent of anticipated take, coupled with other effects of the proposed action, is not likely to result in jeopardy to the species or destruction or adverse modification of critical habitat.

9.3 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 term and conditions to implement the measures, must be provided. Only incidental take resulting from the agency actions and any specified reasonable and prudent measures and terms and conditions identified in the incidental take statement are exempt from the taking prohibition of section 9(a), pursuant to section 7(o) of the ESA.

“Reasonable and prudent measures” are nondiscretionary measures to minimize the amount or extent of incidental take (50 CFR 402.02). NMFS believes the reasonable and prudent measures described below are necessary and appropriate to minimize the impacts of incidental take on threatened and endangered species:

(1) NMFS’ Permits and Conservation Division shall require the U.S. Navy to implement a program to mitigate the potential effects of SURTASS LFA sonar transmissions on threatened or endangered species as specified in the final regulations for the Taking of Marine Mammals Incidental to Operation of the Surveillance Towed Array Sensor System Low Frequency Active Sonar (50 CFR § 218 Subpart X).

(2) NMFS’ Permits and Conservation Division shall require the U.S. Navy to implement a program to monitor potential interactions between SURTASS LFA sonar transmissions and threatened or endangered species.

9.4 Terms and Conditions

To be exempt from the prohibitions of section 9 of the ESA, the Permits and Conservation Division and 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 Permits and Conservation Division and the U.S. Navy fail to ensure compliance with these terms and conditions and their implementing reasonable and prudent measures, the protective coverage of section 7(o)(2) may lapse.

(1) The authorization shall be valid only for the activities associated with the operation of the SURTASS LFA sonar onboard the USNS *IMPECCABLE* (T-AGOS 23),

USNS *ABLE* (T-AGOS 20), USNS *EFFECTIVE* (T-AGOS 21) and USNS *VICTORIOUS* (T-AGOS 19). The signals transmitted by the SURTASS LFA sonar source must be between 100 and 500 Hz with a source level for each projector no more than 215 dB re: 1 µPa at 1 m and a maximum duty cycle of 20 percent.

(2) The U.S. Navy shall be required to (a) establish shut-down criteria for the SURTASS LFA sonar whenever a marine mammal or other ESA-listed species (i.e., sea turtles, scalloped hammerhead shark, Chinese and Sakhalin sturgeon) is detected within the one km (0.54 nmi) buffer zone beyond the SURTASS LFA sonar mitigation zone (180 dB sound field), (b) not broadcast the SURTASS LFA sonar signal at a frequency greater than 500 Hz, and (c) plan its missions to ensure take levels analyzed in this opinion and conference report are not exceeded.

(3) If a marine mammal or other ESA-listed species is detected within the area subjected to a sound pressure level of 180 dB or greater (mitigation zone) or within the one km (0.5 nmi) buffer zone extending beyond the 180 dB mitigation zone, SURTASS LFA sonar transmissions shall be immediately delayed or suspended. Transmissions shall not resume earlier than 15 minutes after:

  a. All marine mammals or other ESA-listed species have left the area of the LFA mitigation and buffer zone; and

  b. There is no further detection of any marine mammal or other ESA-listed species within the LFA mitigation and buffer zones as determined by the visual and/or passive or active acoustic monitoring.

(4) The High Frequency Marine Mammal Monitoring (HF/M3) sonar source described in 50 CFR § 218.235 shall be ramped-up to operating levels over a period of no less than 5 minutes. The HF/M3 source level shall not be increased if a marine mammal or other ESA listed species is detected during ramp-up. The HF/M3 ramp-up may continue once marine mammals or other ESA-listed species are no longer detected. The HF/M3 sonar shall be ramped-up:

  a. At least 30 minutes prior to any SURTASS LFA sonar transmissions;

  b. Prior to any SURTASS LFA sonar calibrations or testing that are not part of regular SURTASS LFA sonar transmissions described in 50 CFR § 218.230; and

  c. Any time after the HF/M3 source has been powered down for more than two minutes.

(5) The SURTASS LFA sonar shall not be operated such that the SURTASS LFA sonar sound field exceeds 180 dB (re: 1µPa [rms]):

  a. Within 22 km (12 nmi) from any coastline, including offshore islands;

b. Within one km (0.5 nmi) seaward of the outer perimeter of any designated OBIA (i.e., the Hawaiian Islands Humpback Whale National Marine Sanctuary—Penguin Bank OBIA (OBIA #16) located in the north-central Pacific Ocean) during the biologically important season for that particular area in accordance with 50 CFR § 218.230.

(6) The U.S. Navy shall deliver an annual report no later than 45 days after the expiration of any LOA issued for the operation of SURTASS LFA sonar. This report shall include numbers and locations of threatened and endangered species sightings, and all information required by the LOA, including the results, if any, of coordination with coastal marine mammal stranding networks. The annual reports shall be submitted to the following NMFS offices: (1) Chief, Permits and Conservation Division, 1315 East-West Highway, Silver Spring, Maryland; and (2) Chief, ESA Interagency Cooperation Division, 1315 East-West Highway, Silver Spring, Maryland.

(7) The U.S. Navy shall collect specific data on any apparent avoidance reactions of threatened or endangered species in response to exposure to SURTASS LFA sonar transmissions, including the distance from the LFA sonar transmission, conditions of the exposure (location coordinates, depth of the species, time of day, ocean conditions, the animal's behavior before and after the exposure, and estimates of the received levels that elicited the response). These data must be reported in the annual reports described in condition 6 (above).

(8) If the U.S. Navy's monitoring programs identify any threatened or endangered species that demonstrate acute effects in response to exposure to LFA sonar transmissions, such as injury or death, the U.S. Navy shall immediately initiate the source shut-down protocol for the sonar system.

(9) The U.S. Navy shall carry out all mitigation, monitoring and reporting requirements contained in the LOAs issued under section 101(a)(5)(A) of the MMPA.

(10) Systematically observe SURTASS LFA sonar operations for injured or disabled marine mammals and monitor the principal marine mammal stranding networks and other media to correlate analysis of any whale strandings that could potentially be associated with SURTASS LFA sonar operations.

These reasonable and prudent measures, with their implementing terms and conditions, are designed to minimize the impact of incidental take that might otherwise result from the proposed action. If, during the course of the action, the level of incidental take specified in this incidental take statement is exceeded, NMFS' Permits and Conservation Division and the U.S. Navy must immediately reinitiate consultation and review the reasonable and prudent measures provided. NMFS' Permits and Conservation Division and U.S. Navy must immediately provide an explanation, in writing, of the causes of any take and discuss possible modifications to the reasonable and prudent measures with NMFS' ESA Interagency Cooperation Division.

10 CONSERVATION RECOMMENDATIONS

Section 7(a)(1) of the ESA directs Federal agencies to use their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of the threatened and endangered species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on ESA-listed species or designated critical habitat, to implement recovery plans or develop information (50 CFR § 402.02).

1. We recommend SURTASS LFA sonar environmental compliance efforts integrate more effectively with U.S. Navy fleet training and testing environmental compliance efforts, including the development and implementation of acoustic threshold criteria in modeling efforts, the adaptive management process, and literature review to ensure consistency and to explore efficiencies pertaining to ESA Section 7(a)(2).

2. We recommend the U.S. Navy pursue research to determine the effectiveness of the HF/M3 active sonar system at detecting ESA-listed sea turtles and fish, including sturgeon and scalloped hammerhead sharks.

3. We recommend the U.S. Navy pursue updated research to determine the potential effects of LFA sonar on marine mammals and sea turtles.

4. We recommend the U.S. Navy consider requesting formal consultation to assess SURTASS LFA sonar activities over a longer period of time to the extent that activity levels and locations are reasonably foreseeable. This may or may not require requesting multi-year LOAs within the five-year period of MMPA regulations.

In order for NMFS' ESA Interagency Cooperation Division to be kept informed of actions minimizing or avoiding adverse effects on, or benefiting, ESA-listed species or their designated critical habitat, the Permits and Conservation Division should notify the ESA Interagency Cooperation Division of any conservation recommendations they implement in their final action.

11 REINITIATION OF CONSULTATION

This concludes formal consultation on the U.S. Navy's proposed use of SURTASS LFA sonar and NMFS' proposed issuance of four Letters of Authorization that would allow the U.S. Navy to "take" marine mammals incidental to its employment of the SURTASS LFA sonar system from August 15, 2016 through August 14, 2017, pursuant to the provisions of section 7 of the ESA and MMPA. As provided in 50 CFR § 402.16, reinitiation of formal consultation is required where discretionary Federal agency involvement or control over the action has been retained (or is authorized by law) and if: (1) the amount or extent of incidental take is exceeded; (2) new information reveals effects of the agency action that may affect ESA-listed species or designated critical habitat in a manner or to an extent not considered in this opinion and conference report; (3) the agency action is subsequently modified in a manner that causes an effect to the ESA-listed species or designated critical habitat not considered in this opinion and conference report; or (4) a new species is ESA-listed or critical habitat designated that may be affected by the action.

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