NOAA's National Marine Fisheries Service
Endangered Species Act Section 7 Consultation

Biological Opinion

Agencies:	United States Geological Survey And The Permits, Conservation and Education Division of the Office of Protected Resources, NOAA's National Marine Fisheries Service
Activities Considered:	Geophysical Survey by the United States Geological Survey in the Gulf of Alaska and Issuance of an Incidental Harassment Authorization Pursuant to Section 101(a)(5)(D) of the Marine Mammal Protection Act (MMPA)
Consultation Conducted by:	Endangered Species Division of the Office of Protected Resources, NOAA's National Marine Fisheries Service
Approved by:	[illegible] for J. H. Lecky
Date:	JUN 0 3 2011

Section 7(a)(2) of the Endangered Species Act (ESA) (16 U.S.C. 1531 et seq.) requires that each federal agency shall ensure that any action authorized, funded, or carried out by such agency is not likely to jeopardize the continued existence of any endangered or threatened species or result in the destruction or adverse modification of critical habitat of such species. When the action of a federal agency "may affect" a listed species or critical habitat that has been designated for such species, that agency is required to consult with either NOAA's National Marine Fisheries Service (NMFS) or the U.S. Fish and Wildlife Service, depending upon the listed resources that may be affected.

This ESA section 7 consultation consideres two proposed actions. The first action is the United States Geological Survey's (USGS') proposed conduct of a geophysical survey in the Gulf of Alaska. As the conduct of this survey would result in takes of marine mammals incidental to this survey, USGS must also apply for an Incidental Harassment Authorization (IHA) under the Marine Mammal Protection Act (MMPA), 16 U.S.C. §1371 (a)(5)(D); therefore, the NMFS' Office of Protected Resources Permits, Conservation and Education Division (Permits Division) is proposing to issue an IHA. The issuance of the IHA is the second proposed action. Subsequently, the action agencies for this consultation are USGS and the Permits Division. The consulting agency is the Office of Protected Resources, Endangered Species Division.

This document represents NMFS' biological opinion (Opinion) on the effects of the proposed actions on endangered and threatened species and designated critical habitat, prepared in accordance with section 7 of the ESA. This Opinion is based on information provided in the IHA application, draft IHA, environmental assessment, monitoring reports from similar activities, published and unpublished scientific information on endangered and threatened species, scientific and commercial information such as reports from government agencies and the peer-reviewed literature, biological opinions on similar activities, and other sources of information.

Consultation History

On January 21, 2011, USGS submitted an application for an IHA from the Permits Division for the Gulf of Alaska survey. This request was accompanied by an environmental assessment regarding the potential impacts of the proposed suvey on the environment.

On February 3, 2011, USGS requested formal consultation on their proposed conduct of a geophysical survey. This survey would be conducted from June 5 through June 25, 2011.

On March 26, 2011, F/PR3 received a request for initiation of consultation from the Permits Division regarding the proposed issuance of an IHA for harassment of marine mammals incidental to the proposed survey.

F/PR3 initiated consultation on the proposed survey and IHA on March 26, 2011.

On May 13, 2011, the Permits Division provided a draft IHA to the Endangered Species Division. On May 25, 2011, a revised draft IHA was provided to the Endangered Species Division.

BIOLOGICAL OPINION

Description of the Proposed Actions

The proposed action for this consultation involves two related actions. USGS proposes to conduct a marine geophysical survey using acoustic sources deployed from the R/V Langseth within the central Gulf of Alaska. The Lamont-Doherty Earth Observatory (L-DEO) will operate the Langseth as well as direct the operation of all acoustic sources for USGS. As this survey is expected to incidentally harass marine mammals USGS has also requested an IHA. The Permits Division proposes to issue an IHA pursuant to Section 101(a)(5)(D) of the Marine Mammal Protection Act, 16 U.S.C. 1371 (a)(5)(D) to USGS. The IHA would be valid from June 5 through July 25, 2011, and would authorize the incidental harassment of fin whales, humpback whales, sperm whales and steller sea lions as well as other non-ESA listed whales and pinnipeds. The Permits Division estimates that the proposed action would incidentally harass 76 fin whales, 68 humpback whales and 10 sperm whales. Listed sea turtles would also be harassed during the conduct of the seismic activities.

Overview of the Seismic Survey
All acoustic sources will be deployed from the *R/V Marcus G. Langseth*. These acoustic sources are comprised of a 36-airgun array, a multibeam ecosounder (MBES), a subbottom profiler (SBP) and ocean bottom seismometers (OBSs). The MBES and the SBP would operate continuously with the airgun array throughout the entire cruise. The *R/V Langseth* would tow a hydrophone streamer to receive the returning acoustic signals. This survey also requires the deployment and retrieval of OBSs which would record returning acoustic signals for later analysis. The purpose of this survey is to collect seismic reflection and refraction profiles for delineating the U.S. extended continental shelf in the Gulf of Alaska.

The *R/V Langseth* is expected to depart Dutch Harbor, Alaska around June 5, 2011and return to Dutch Harbor around July 25, 2011. Transit to the survey site in the central Gulf of Alaska will take approximately two days. The survey schedule is as follows:

- Reflection survey: approximately 10 days
- OBS deployment: 1 day
- Refraction Survey: approximately 2 days
- OBS recovery: approximately 2 days
- Contingency survey lines (if time permits): approximately 2 days

The survey will consist of about 2,840 kilometer (km) of transect lines within the Gulf of Alaska with an additional 140 km of turns. Additional seismic acquisition may occur along a 340-km contingency line if time permits. These line estimates include turns the vessel will make along the survey and additional operations associated with equipment testing, startup, line changes, and repeat coverage as necessary.

Source Vessel, Support Vessels and Equipment
The *Langseth* has a length of 71.5 m, a beam of 17.0 m, and a maximum draft of 5.9 m. It is the sole source vessel for this survey. The *Langseth* was designed with a propulsion system designed to be as quiet as possible to avoid interference with the seismic signals. The ship is powered by two Bergen BRG-6 diesel engines, each producing 3550 hp, which drive the two propellers directly. Each propeller has four blades, and the shaft typically rotates at 750 revolutions per minute (rpm). Althought the vessel is equipped with an 800 hp bowthruster, it will not be used during seismic acquisition. Typical cruising speed for the Langseth is 18.5 km/hour but during seismic acquisition will travel at 7.4–9.3 km/hour. As the vessel tows the airgun array and streamer the turning rate and maneuverability of the vessel will be limited.

Airguns
Although a two string, 3300 in^3 array may be used if possible, USGS expects that a 36-airgun array would be used during the proposed study, therefore, the larger 36-airgun array will be described here. This array is comprised of a mixture of Bolt 1500LL and 1900LL (2,000 psi) airguns, each ranging in size from 40–360 in^3 as appropriate. Airguns would be arranged in a configuration of four identical linear strings with a total discharge volume of 6,600 in^3. Energy for the airgun array would be compressed air supplied by compressors on board the *Langseth*. Each string contains 10 airguns, with 9 of them to be fired simultaneously and 1 reserved as a spare. The first and last airguns in each string would be spaced 16 m (52 ft) apart, with the four

strings distributed across an area of approximately 24 m x 16 m (79 ft x 52 ft). The array would be towed about 100 m (328 ft) behind the *Langseth* at a 9 m (29.5 ft) depth.

Airguns would fire for a brief (0.1 second) pulse[1] at intervals of every 22 s (about every 50 m) for seismic surveying with the hydrophone streamer, or every 66 s (about every 150 m) when recording data on the OBSs. The airguns would be silent during intervening periods. The airgun array is predominantly low-frequency[2] with a dominant frequency component of 0–188 Hz.

Source output (downward) from the 36-airgun array would be 259 dB re 1 μPa m (0-pk) and 265 dB re 1 μPa • m (pk-pk). The nominal source levels of the array at various tow depths are nearly identical. Because the actual source is a distributed sound source (36 airguns) rather than a single point source, the highest sound levels measurable at any location in the water will be less than the nominal source level.

Receiving System

The receiving system will consist of a hydrophone streamer and five OBSs. The *Langseth* will deploy an 8-km long streamer. As the airgun array is towed along the survey lines, the hydrophone streamer will receive the returning acoustic signals and transfer the data to the on-board processing system.

The OBSs record the returning acoustic signals for later analysis. After the rerfraction survey the OBSs will be retrieved. An acoustic release transponder interrogates the OBS at a frequency of 9–11 kHz, and a response is received at a frequency of 9–13 kHz. The burn wire release assembly is then activated, and the instrument is released from the anchor to float to the surface. All OBSs will be retrieved at the end of the study.

Multi-beam Bathymetric Echosounder (MBES)

The proposed study would use a MBES to map the ocean floor. The MBES sonar would be operated from the *Langseth* simultaneously with the airgun array. The hull-mounted Kongsberg EM 122 would operate at 10.5-13 kHz (usually 12 kHz), and have a maximum source level of 242 dB re 1 μPa (rms). The transmitting beamwidth is 1 or 2° fore-aft and 150° athwartship.

For deep-water operation, each “ping” consists of eight (in water depths greater than 1000 m) or four (in water depths less than 1000 m) successive fan-shaped transmissions which ensonifies a sector that extends 1° fore-aft. The successive transmissions span an overall cross-track angular extent of about 150°, with 2-ms gaps between pings for successive sectors. Continuous wave signals (waves of constant energy and frequency) increase from 2 to 15 ms long in water depths up to 2600 m and frequency-modulated (FM) chirp signals up to 100 ms long are used in water depths greater than 2600 m.

Sub-Bottom Profiler (SBP)

The proposed survey would also use a Knudsen 320B sub-bottom profiler to provide information about the sea floor. The SBP would operate simultaneously with the airgun array and the MBES.

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1 Pulse duration is defined as the time at which 5 percent to 95 percent of the pulse energy has arrived.

2 Frequencies are categorized as low-frequency (< 1,000 Hz), mid-frequency (1-10 kHz), and high-frequency (> 10 kHz).

Energy from the SBP is directed downward by a hull-mounted 3.5 kHz transducer and varies with water depth. The SBP has a maximum source level of 204 dB re 1 μPa but varies with water depth. Nominal beam width is 30 degrees, and pulse duration would be 1, 2, or 4 ms, with bandwidths of 1.0 kHz, 0.5 kHz, and 0.25 kHz, respectively. The interval between SBP pulses would be 1 s, with a common mode of operation being five pulses at 1-s intervals followed by a 5-s pause.

Monitoring and Reporting

USGS proposes to document the nature and extent of any effects on listed species through the use of observers, monitoring efforts and reporting. Vessel-based Protected Species Visual Observers (PSVOs) on the *Langseth* would watch for marine mammals and sea turtles near seismic sources and the vessel during all daytime airgun operations, as well as during any night-time start-ups of the array. PSVOs would also watch for marine mammals and turtles near the vessel for at least 30 min prior to the planned start of seismic operations after an extended shut-down of the airguns. When feasible, observations would also be made during daytime periods without seismic operations (e.g., during transits). When marine mammals or sea turtles are observed within, or about to enter, designated exclusion zones [See *Exclusion Zone* (EZ) section below], airgun operations would immediately be powered-down (or shut-down if necessary). Airgun operations would not resume until the animal(s) leaves the EZ, as determined by the PSVOs.

Although at least four PSVOs will be based aboard the Langseth, three PSVOs are typically on watch at a time, two on the observation tower conducting visual observations and the third monitoring the PAM equipment. On the tower, two observers are on watch during all daylight hours except during meal times when one observer may be on watch. Observers typically observe for no more than four hours. The crew of the *Langseth* would also be instructed to assist in detecting marine mammals and turtles as well as implementing mitigation measures as possible.

During daytime, the PSVOs would systematically scan the area around the vessel with reticle binoculars (e.g., 7×50 Fujinon), Big-eye binoculars (25×150), and the naked eye. Laser rangefinding binoculars (Leica LRF 1200 laser rangefinder or equivalent) would be available to assist with distance estimation. At night, the PSVOs would use night vision devices when required. Laser rangefinding binoculars will be available to assist with visual distance estimation. When stationed on the observation platform on the *Langseth*, eye level is about 21.5 m (70.5 ft) above sea level and PSVOs would be able to see around the entire vessel and to a distance of about 10 km with the naked eye, 5 km or further with the big eyes, 200 m with night vision and 2-3 km or further with the reticle binoculars.

Passive Acoustic Monitoring (PAM)

Passive acoustic monitoring would also be conducted by the *Langseth* during all proposed seismic operations and during most periods when underway and when the airguns are not operating. Protected Species Acoustic Observers (PSAOs) would monitore PAM in real-time 24 hours per day as practical during daytime and nighttime operations to alert visual observers when vocalizing cetaceans are detected. Any bearings to cetaceans determined using PAM would be relayed to the visual observer.

The PAM system that would be used consists of a low-noise, towed hydrophone array connected to the vessel by a 'hairy' faired cable. The array would be deployed from a winch on the back deck of the *Langseth*. The lead-in from the hydrophone array is approximately 400 m (1,312 ft) in length, and the active part of the hydrophone array is approximately 56 m (184 ft) long. The PAM hydrophone array is typically towed at depths of less than 20 m (66 ft).

One PSAO would monitor the PAM system by listening to the signals from two channels via headphones and/or speakers and watching real-time spectrographic display for frequency ranges produced by cetaceans. Shifts for PSAOs monitoring the acoustical data would range from one to six hours in length, with all PSVOs expected to rotate through the PAM position. When a vocalization is detected, the acoustic PSAO would contact the PSVO immediately to alert the visual observer of the vocalizing animal(s) in case a power-down or shut-down is required. The theoretical distance for cetacean call detection by PAM is in the tens of kilometers, but detection is dependent on several factors including call intensity, ship noise, ambient noise in the water column, and physical/oceanographic conditions.

A report will be submitted to NMFS within 90 days after the end of the cruise. The report will describe the operations that were conducted and sightings of marine mammals and turtles near the operations. The report will provide full documentation of methods, results, and interpretations of monitoring efforts. The 90-day report will summarize the dates and locations of seismic operations, and all marine mammal and turtle sightings (dates, times, locations, activities, associated seismic survey activities). The report will also include estimates of the amount and nature of any potential "take" of marine mammals and sea turtles by harassment or in other ways.

Mitigation

USGS has adopted mitigation measures to minimize adverse affects to listed species. These measures include: (1) airgun power-down and shut-down procedures, emergency shut-down and ramp-up procedures within calculated exclusion zones; and (2) immediate shut-down when blue, sei and North Pacific right whales and concentrations of humpback and fin whales are sighted.

Proposed Exclusion Zones for Power Down and Shut Down Procedures

L-DEO predicted the propagation of sound from the airgun array and from a single 1900LL 40-in^3 airgun, which will be used during power downs during the proposed survey. Table 1 provides the estimated distances for propagation radii at 160, 170, 180, and 190 dB re 1μPa (rms) in deep water (greater than 1,000 m, 3,281 ft) as the survey will occur in deep water in the Gulf of Alaska. The 180 dB re 1μPa (rms) distance represents the exclusion zone criterion and the thresholds for power-down or shut down for cetaceans and sea turtles during the proposed survey (see *Mitigation and Monitoring* section below). Estimated propagation distances to the 180 dB isopleth range from 40 m (0.02 nm) to 940 m (0.51 nm) depending on seismic source and volume (se Table 1 below). The 190 dB re 1μPa (rms) distance represents the exclusion zone criterion and the thresholds for power-down or shut down for pinnipeds during the proposed survey. Estimated propagation distances to the 190 dB isopleth range from 12 m (0.01 nm) to 400 m (0.22 nm) depending on seismic source and volume.

Empirical propagation measurements for the Langseth's airguns were taken during a calibration study in 2007-2008 in the Gulf of Mexico (Tolstoy et al 2009). For the study the 36-airgun array was towed at 6 m (19.6 ft) in deep water (~1,600 m; 5,249 ft) as well as shallow water (~50 m; 131 ft). L-DEO modeled received level radii and compared the model results with the Langseth's calibration results in shallow, intermediate, and deep water. The model represented the actual produced received levels, particularly within the first few kms, where the predicted exclusion zones (EZs) (i.e., safety radii) lie. At greater distances, local oceanographic variations begin to take effect, and the model tends to over predict received levels. Since the modeling matches the observed measurements, the model continues to be used for defining EZs and mitigation radii for various tow depths. Although the L-DEO model does not account for site-specific environmental conditions the calibration study of the L-DEO model predicted that using site-specific information may actually provide less conservative EZ radii at greater distances.

Table 1. Distances to which sound levels ≥ 190, 180, 170, and 160 dB re 1 μPa_{rms} could be received from the single airgun and the airgun array.

Source and Volume	Tow Depth (m)	Water Depth*	Estimated RMS Radii (m)			
			190 dB	180 dB	170 dB	160 dB
Single Bolt airgun (40 in^3)	9	Deep	12	40	120	385
4 strings 36 airguns (6600 in^3)	9	Deep	400	940	2200	3850

Power Down and Shut Down Procedures

If a marine mammal or turtle is detected outside the 180 dB (rms) EZ but is likely to enter it, USGS would power-down the airgun array before the animal is within the EZ. If a mammal or turtle is already within the EZ when first detected, the airguns would be powered-down immediately. A power-down may also occur when the vessel is moving from one track line to another (i.e., during a turn). Power-down procedures involve reducing the number of operating airguns, typically to a single airgun (e.g., 40 in^3), to minimize the EZ so that marine mammals or turtles are no longer in or about to enter the 180 dB radii. The continued operation of at least one airgun during a power-down is intended to alert marine mammals and turtles to the presence of the seismic vessel in that area. Criteria for airgun shut down procedures (i.e., all operting airguns are turned off) include: if, during operation of the single airgun (as in during power down), a marine mammal or turtle is detected within the EZ, or if during seismic activity a blue, fin, sei, North Pacific right whale or concentrations of humpback and sperm whales is observed regardless of the distance to the vessel and EZ. Ramp up procedures would only take place if these whales are not seen for 30 minutes.

Following a power-down or shut-down, airgun activity would not resume until the marine mammal or turtle has cleared the EZ, or until the PSVO is confident the animal has left the vicinity of the vessel. This is considered to have occurred if the animal:

- is visually observed to have left the EZ, *or*
- has not been seen within the zone for 30 minutes (in the case of mysticetes and large odontocetes, including sperm whales) or 15 minutes (in the case of pinnipeds), *or*
- the vessel has moved outside the EZ for sea turtles (8 min of travel: the time it would take the Langseth to move outside the 180-dB EZ).

Ramp-up Procedures

Following a power-down or shut-down and subsequent clearing of the EZ, the airgun array would resume operations according to the following procedures.

If after eight minutes with either no airgun operations or a power-down of airguns, USGS will implement ramp-up procedures for the array. This eight minute period is based on the time it would take the Langseth to move outside the 180-dB EZ for mammals and turtles. Ramp up from a state of no airgun operations would begin with the smallest airgun in the array (40 in^3). Airguns would be added in a sequence such that the increase in source level would not exceed 6 dB/5min over a total duration of about 35 minutes. Ramp up from a reduced power state, such as during maintenance of an airgun string while the remaining strings keep firing, would include the start-up of the returned string. During ramp-up, the PSVOs would monitor the EZ for the full airgun array, and if marine mammals or turtles are sighted then power-down, or shut-down would be implemented.

Initiation of ramp-up procedures from a shut-down requires that the full EZ be visible by the PSVOs, whether the ramp-up is conducted in daytime or nighttime. Thus, the airgun array would likely not be ramped-up from a complete shut-down at night or in thick fog, because the outer part of the EZ for the array may not be visible during those conditions. Ramp-up of airguns would be allowed under reduced visibility conditions only if at least one airgun (e.g., 40 in^3 or similar) has operated continuously, on the assumption that marine mammals and turtles would be alerted to the approaching seismic vessel by sounds from the single airgun and could move away. Ramp up of the airguns would not be initiated if a sea turtle or marine mammal is sighted within or near the applicable EZ during the day or near the vessel at night.

Marine Mammal Protection Act Incidental Harassment Authorization

The Permits Division has also proposed to issue an IHA to USGS for the harassment of marine mammals incidental to the seismic surveys. This IHA is valid from June 5, 2011 to July 25, 2011. The proposed IHA includes the requirements USGS must comply with as part of its authorization. Following are the sections of the IHA that are relevant to listed species:

1. <u>Mitigation and Monitoring Requirements</u>

   USGS is required to implement the following mitigation and monitoring requirements when conducting this suvey to achieve the least practicable adverse impact on affected marine mammal species or stocks:

(a) Utilize two, NMFS-qualified, vessel-based Protected Species Visual Observers (PSVOs) (except during meal times and restroom breaks, when at least one PSVO shall be on watch) to visually watch for and monitor marine mammals near the seismic source vessel during daytime airgun operations (from nautical twilight-dawn to nautical twilight-dusk) and before and during start-ups of airguns day or night. The *Langseth*'s vessel crew shall also assist in detecting marine mammals, when practicable. PSVOs shall have access to reticle binoculars (7x50 Fujinon), big-eye binoculars (25x150), and night vision devices. PSVO shifts shall last no longer than 4 hours at a time. PSVOs shall also make observations during daytime periods when the seismic system is not operating for comparison of animal abundance and behavior, when feasible.

(b) PSVOs shall conduct monitoring while the airgun array and streamer(s) are being deployed or recovered from the water.

(c) Record the following information when a marine mammal is sighted:

> (i) species, group size, age/size/sex categories (if determinable), behavior when first sighted and after initial sighting, heading (if consistent), bearing and distance from seismic vessel, sighting cue, apparent reaction to the airguns or vessel (e.g., none, avoidance, approach, paralleling, etc., and including responses to ramp-up), and behavioral pace; and
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> (ii) time, location, heading, speed, activity of the vessel (including number of airguns operating and whether in state of ramp-up or power-down), Beaufort sea state and wind force, visibility, and sun glare; and
>
> (iii) the data listed under Condition 7(c)(ii) shall also be recorded at the start and end of each observation watch and during a watch whenever there is a change in one or more of the variables.

(d) Utilize the passive acoustic monitoring (PAM) system, to the maximum extent practicable, to detect and allow some localization of marine mammals around the *Langseth* during all airgun operations and during most periods when airguns are not operating. One NMFS-qualified Protected Species Observer (PSO) and/or expert bioacoustician (i.e., Protected Species Acoustic Observer [PSAO]) shall monitor the PAM at all times in shifts no longer than 6 hours. An expert bioacoustician shall design and set up the PAM system and be present to operate or oversee PAM, and available when technical issues occur during the survey.

(e) Do and record the following when an animal is detected by the PAM:

> (i) notify the on-duty PSVO(s) immediately of a vocalizing marine mammal so a power-down or shut-down can be initiated, if required;

(ii) enter the information regarding the vocalization into a database. The data to be entered include an acoustic encounter identification number, whether it was linked with a visual sighting, date, time when first and last heard and whenever any additional information was recorded, position, and water depth when first detected, bearing if determinable, species or species group (e.g., unidentified dolphin, sperm whale), types and nature of sounds heard (e.g., clicks, continuous, sporadic, whistles, creaks, burst pulses, strength of signal, etc.), and any other notable information.

(f) Visually observe the entire extent of the exclusion zone (EZ) (180 dB re 1 μPa [rms] for cetaceans and 190 dB re 1 μPa [rms] for pinnipeds using NMFS-qualified PSVOs, for at least 30 minutes (min) prior to starting the airgun array (day or night). If the PSVO finds a marine mammal within the EZ, USGS must delay the seismic survey until the marine mammal(s) has left the area. If the PSVO sees a marine mammal that surfaces, then dives below the surface, the PSVO shall wait 30 min. If the PSVO sees no marine mammals during that time, they should assume that the animal has moved beyond the EZ. If for any reason the entire radius cannot be seen for the entire 30 min (i.e., rough seas, fog, darkness), or if marine mammals are near, approaching, or in the EZ, the airguns may not be ramped-up. If one airgun is already running at a source level of at least 180 dB re 1 μPa (rms), USGS may start the second airgun without observing the entire EZ for 30 min prior, provided no marine mammals are known to be near the EZ (in accordance with Condition 7[h] below).

(g) Establish a 180 dB re 1 μPa (rms) and 190 dB re 1 μPa (rms) EZ for marine mammals before the 4-string airgun array (6,600 in^3) is in operation; and a 180 dB re 1 μPa (rms) and 190 dB re 1 μPa (rms) EZ before a single airgun (40 in^3) is in operation, respectively. See Table 1 (attached) for distances and EZs.

(h) Implement a "ramp-up" procedure when starting up at the beginning of seismic operations or anytime after the entire array has been shutdown for more than 8 min, which means start the smallest gun first and add airguns in a sequence such that the source level of the array shall increase in steps not exceeding approximately 6 dB per 5-min period. During ramp-up, the PSVOs shall monitor the EZ, and if marine mammals are sighted, a power-down, or shut-down shall be implemented as though the full array were operational. Therefore, initiation of ramp-up procedures from shut-down requires that the PSVOs be able to view the full EZ as described in Condition 7(f) (above).

(i) Alter speed or course during seismic operations if a marine mammal, based on its position and relative motion, appears likely to enter the relevant EZ. If speed or course alteration is not safe or practicable, or if after alteration the marine mammal still appears likely to enter the EZ, further mitigation measures, such as a power-down or shut-down, shall be taken.

(j) Power-down or shut-down the airgun(s) if a marine mammal is detected within, approaches, or enters the relevant EZ. A shut-down means all operating airguns are shut-down (i.e., turned off). A power-down means reducing the number of operating airguns

to a single operating 40 in^3 airgun, which reduces the EZ to the degree that the animal(s) is no longer in or about to enter it.

(k) Following a power-down, if the marine mammal approaches the smaller designated EZ, the airguns must then be completely shut-down. Airgun activity shall not resume until the PSVO has visually observed the marine mammal(s) exiting the EZ and is not likely to return, or has not been seen within the EZ for 15 min for species with shorter dive durations (small odontocetes and pinnipeds) or 30 min for species with longer dive durations (mysticetes and large odontocetes, including sperm, pygmy sperm, dwarf sperm, killer, and beaked whales).

(l) Following a power-down or shut-down and subsequent animal departure, airgun operations may resume following ramp-up procedures described in Condition 1(h).

(m) Marine geophysical surveys may continue into night and low-light hours if such segment(s) of the survey is initiated when the entire relevant EZs are visible and can be effectively monitored.

(n) No initiation of airgun array operations is permitted from a shut-down position at night or during low-light hours (such as in dense fog or heavy rain) when the entire relevant EZ cannot be effectively monitored by the PSVO(s) on duty.

(o) If a North Pacific right (*Eubalaena japonica*), sei (*Balaenoptera borealis*), blue (*Balaenoptera musculus*), and/or beluga whale (*Delphinapterus leucas*) is visually sighted, the airgun array shall be shut-down regardless of the distance of the animal(s) to the sound source. The array shall not resume firing until 30 min after the last documented whale visual sighting.

(p) Concentrations of humpback (*Megaptera novaeangliae*), fin (*Balaenoptera physalus*), and/or killer (*Orcinus orca*) whales shall be avoided, if possible, and the array shall be powered-down if necessary. A concentration or group of whales shall consist of when three or more individuals are visually sighted and do not appear to be traveling (e.g., feeding, socializing, etc.).

(q) To the maximum extent practicable, schedule seismic operations (i.e., shooting airguns) during daylight hours and OBS operations (i.e., deploy/retrieve) to nighttime hours.

2. <u>Reporting Requirements</u>

USGS is required to:

(a) Submit a draft report on all activities and monitoring results to the Office of Protected Resources, NMFS, within 90 days of the completion of the *Langseth*'s central GOA cruise. This report must contain and summarize the following information:

(i) Dates, times, locations, heading, speed, weather, sea conditions (including Beaufort sea state and wind force), and associated activities during all seismic operations and marine mammal sightings;

(ii) Species, number, location, distance from the vessel, and behavior of any marine mammals, as well as associated seismic activity (number of power-downs and shut-downs), observed throughout all monitoring activities.

(iii) An estimate of the number (by species) of marine mammals that: (A) are known to have been exposed to the seismic activity (based on visual observation) at received levels greater than or equal to 160 dB re 1 μPa (rms) and/or 180 dB re 1 μPa (rms) for cetaceans and 190 dB re 1 μPa (rms) for pinnipeds with a discussion of any specific behaviors those individuals exhibited; and (B) may have been exposed (based on reported and corrected empirical values for the 36 airgun array and modeling measurements for the single airgun) to the seismic activity at received levels greater than or equal to 160 dB re 1 μPa (rms) and/or 180 dB re 1 μPa (rms) for cetaceans and 190 dB re 1 μPa (rms) for pinnipeds with a discussion of the nature of the probable consequences of that exposure on the individuals that have been exposed.

(iv) A description of the implementation and effectiveness of the: (A) terms and conditions of the Biological Opinion's Incidental Take Statement (ITS) (attached); and (B) mitigation measures of the Incidental Harassment Authorization. For the Biological Opinion, the report shall confirm the implementation of each Term and Condition, as well as any conservation recommendations, and describe their effectiveness, for minimizing the adverse effects of the action on Endangered Species Act-listed marine mammals.

(b) Submit a final report to the Chief, Permits, Conservation, and Education Division, Office of Protected Resources, NMFS, within 30 days after receiving comments from NMFS on the draft report. If NMFS decides that the draft report needs no comments, the draft report shall be considered to be the final report.

3. In the unanticipated event that the specified activity clearly causes the take of a marine mammal in a manner prohibited by this Authorization, such as an injury (Level A harassment), serious injury or mortality (e.g., ship-strike, gear interaction, and/or entanglement), USGS shall immediately cease the specified activities and immediately report the incident to the Chief of the Permits, Conservation, and Education Division, Office of Protected Resources, NMFS and the Alaska Regional Stranding Coordinators. The report must include the following information:

(a) time, date, and location (latitude/longitude) of the incident; the name and type of vessel involved; the vessel's speed during and leading up to the incident; description of the incident; status of all sound source use in the 24 hours preceding the incident; water depth; environmental conditions (e.g., wind speed and direction, Beaufort sea state, cloud cover, and visibility); description of marine mammal observations in the 24 hours preceding the incident; species identification or description of the animal(s) involved; the

fate of the animal(s); and photographs or video footage of the animal (if equipment is available).

Activities shall not resume until NMFS is able to review the circumstances of the prohibited take. NMFS shall work with USGS to determine what is necessary to minimize the likelihood of further prohibited take and ensure MMPA compliance. USGS may not resume their activities until notified by NMFS via letter or email, or telephone.

In the event that USGS discovers an injured or dead marine mammal, and the lead PSO determines that the cause of the injury or death is unknown and the death is relatively recent (i.e., in less than a moderate state of decomposition as described in the next paragraph), USGS will immediately report the incident to the Chief of the Permits Conservation, and Education Division, Office of Protected Resources, NMFS, at 301-713-2289, and the NMFS Alaska Stranding Hotline (1-877-925-7773) and/or by email to the Alaska Regional Stranding Coordinators. The report must include the same information identified in the paragraph 3(a) above. Activities may continue while NMFS reviews the circumstances of the incident. NMFS will work with USGS to determine whether modifications in the activities are appropriate.

In the event that USGS discovers an injured or dead marine mammal, and the lead PSO determines that the injury or death is not associated with or related to the survey activities authorized in this LOA, USGS shall report the incident to the Chief of the Permits, Conservation, and Education Division, Office of Protected Resources, NMFS, at 301-713-2289, and the NMFS Alaska Stranding Hotline (1-877-925-7773) and/or by email to the Alaska Regional Stranding Coordinators within 24 hours of the discovery. USGS shall provide photographs or video footage (if available) or other documentation of the stranded animal sighting to NMFS and the Marine Mammal Stranding Network.

Approach to the Assessment

NMFS approaches its section 7 analyses of agency actions through a series of steps. The first step identifies those aspects of proposed actions that are likely to have direct and indirect physical, chemical, and biotic effects on listed species or on the physical, chemical, and biotic environment of an action area. As part of this step, we identify the spatial extent of these direct and indirect effects, including changes in that spatial extent over time. The result of this step includes defining the *action area* for the consultation. The second step of our analyses identifies the listed resources that are likely to co-occur with these effects in space and time and the nature of that co-occurrence (these represent our *exposure analyses*). In this step of our analyses, we try to identify the number, age (or life stage), and gender of the individuals that are likely to be exposed to an action's effects and the populations or subpopulations those individuals represent. Once we identify which listed resources are likely to be exposed to an action's effects and the nature of that exposure, we examine the scientific and commercial data available to determine whether and how those listed resources are likely to respond given their exposure (these represent our *response analyses*).

The final steps of our analyses – establishing the risks those responses pose to listed resources – are different for listed species and designated critical habitat (these represent our *risk analyses*). 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 listed, which can include true biological species, subspecies, or distinct population segments of vertebrate species. The continued existence of these "species" depends on the fate of the populations that comprise them. 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, the populations that comprise that species, 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 individual 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 individuals' "fitness," or the individual's growth, survival, annual reproductive success, and lifetime reproductive success. In particular, we examine the scientific and commercial data available to determine if an individual's probable lethal, sub-lethal, or behavioral responses to an action's effect on the environment (which we identify during our response analyses) are likely to have consequences for the individual's fitness.

When individual, listed plants or animals are expected to experience reductions in fitness in response to an action, those fitness reductions are likely to reduce the abundance, reproduction, or growth rates (or increase the variance in these measures) of the populations those individuals represent (see Stearns 1992). Reductions in at least one 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. As a result,

when listed plants or animals exposed to an action's effects are *not* expected 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 (e.g., Brandon 1978; Mills and Beatty 1979; Stearns 1992; Anderson 2000). As a result, if we conclude that listed plants or animals are *not* likely to experience reductions in their fitness, we would conclude our assessment.

Although reductions in fitness of individuals is a *necessary* condition for reductions in a population's viability, reducing the fitness of individuals in a population is not always *sufficient* to reduce the viability of the population(s) those individuals represent. Therefore, if we conclude that listed plants or animals are likely to experience reductions in their fitness, we determine whether those fitness reductions are likely to reduce the viability of the populations the individuals represent (measured using changes in the populations' abundance, reproduction, spatial structure and connectivity, growth rates, variance in these measures, or measures of extinction risk). 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) as our point of reference. If we conclude that reductions in individual fitness are not likely to reduce the viability of the populations those individuals represent, we would conclude our assessment.

Reducing the viability of a population is not always *sufficient* to reduce the viability of the species those populations comprise. Therefore, in the final step of our analyses, we determine if reductions in a population's viability are likely to reduce the viability of the species those populations comprise using changes in a species' reproduction, numbers, distribution, estimates of extinction risk, or probability of being conserved. In this step of our analyses, we use the species' status (established in the *Status of the Species* section of this Opinion) as our point of reference. Our final determinations are based on whether threatened or endangered species are likely to experience reductions in their viability and whether such reductions are likely to be appreciable.

To conduct these analyses, we rely on all of the evidence available to us. This evidence might consist of monitoring reports submitted by past and present permit holders; reports from NMFS Science Centers; reports prepared by natural resource agencies in States, Tribes, and other countries; reports from non-governmental organizations involved in marine conservation issues; the information provided by the Permits, Conservation and Education Division when it initiates formal consultation; and the general scientific literature. We supplement this evidence with reports and other documents – environmental assessments, environmental impact statements, and monitoring reports – prepared by other federal and state agencies like the Minerals Management Service, U.S. Coast Guard, and U.S. Navy whose operations extend into the marine environment.

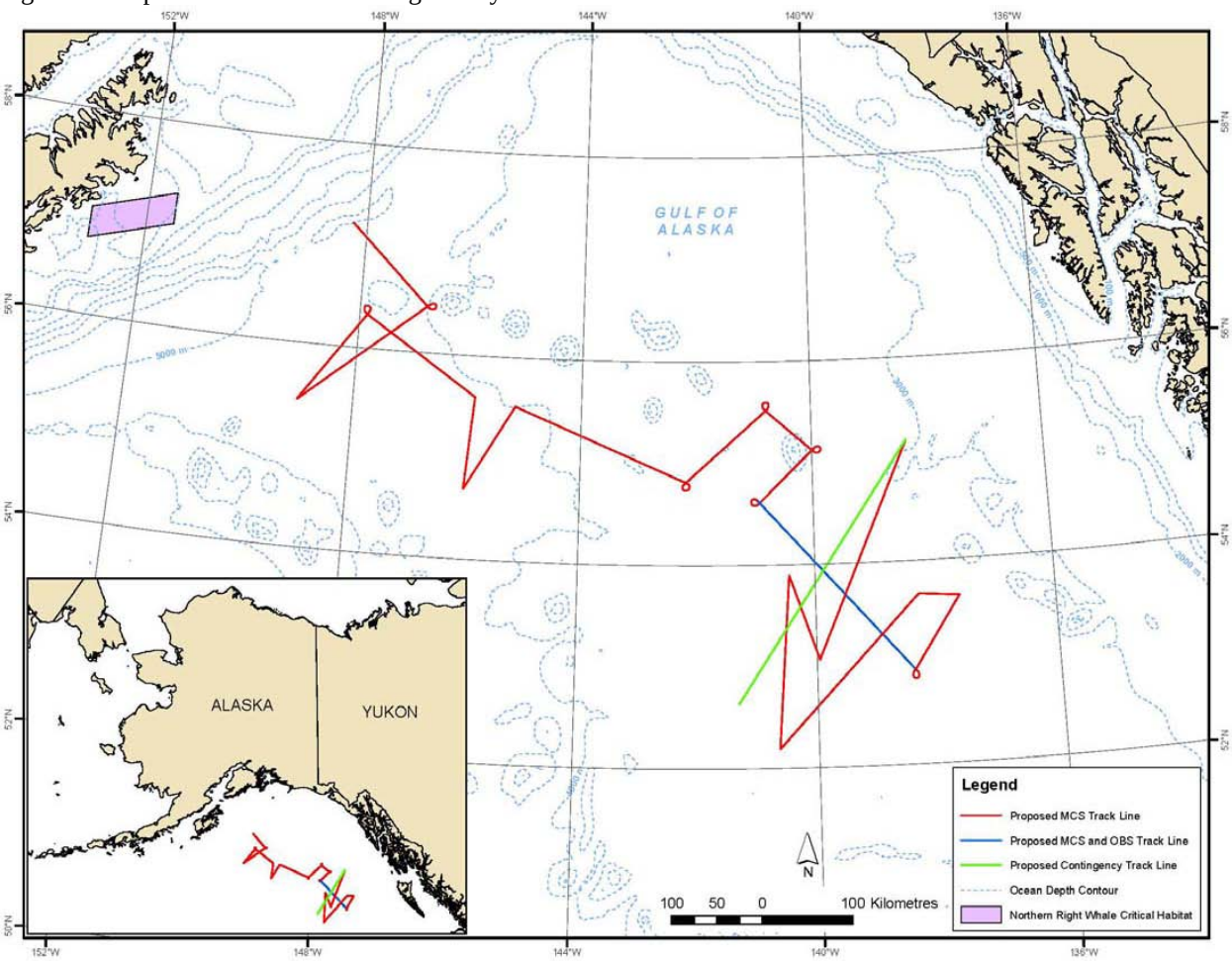
During the consultation, we conducted electronic searches of the general scientific literature using search engines, including *Agricola, Aquatic Sciences and Fisheries Abstracts, Conference Papers Index, Oceanic Abstracts, BioOne, Science Direct, Ingenta Connect, JSTOR, Web of Science - Science Citation Index, First Search (Article First, ECO, WorldCat),* and *Google Scholar*. We supplemented these searches with electronic searches of doctoral dissertations and master's theses. These searches specifically tried to identify data or other information that supports a particular conclusion (for example, a study that suggests whales or turtles will exhibit

a particular response to a seismic source) as well as data that does not support that conclusion. When data were equivocal or when faced with substantial uncertainty, our decisions were designed to avoid the risks of incorrectly concluding that an action would not have an adverse effect on listed species when, in fact, such adverse effects are likely (i.e., Type II error).

Action Area

The action area for this consultation will encompass the survey area delineated by 53° to 57°N, and 135° to 148° W (Fig. 1), approximately 200 to 650 km (108 to 351 nm) from the Alaskan shoreline. Water depths in the survey area range from about 2,000 to more than 6,000 m (6562 to more than 19,685 ft) deep. USGS will survey approximately 3, 300 km of track line and ensonify approximately 26,166 km^2 to 160 dB re 1 μPa (rms).

Figure 1: Map of Action Area Showing Survey Track



Status of Listed Resources

NMFS has determined that the actions considered in this Opinion may affect the following listed resources provided protection under the Endangered Species Act of 1973, as amended (16 U.S.C. 1531 *et seq*.; ESA):

Blue whale	*Balaenoptera musculus*	Endangered
Fin whale	*Balaenoptera physalus*	Endangered
Humpback whale	*Megaptera novaeangliae*	Endangered
North Pacific Right whale	*Eubalaena japonica*	Endangered
Sei whale	*Balaenoptera borealis*	Endangered
Sperm whale	*Physeter macrocephalus*	Endangered
Steller Sea Lion	*Eumetopias jubatus*	
Eastern Distinct Population Segment (DPS)		Threatened
Western DPS		Threatened
Green sea turtle	*Chelonia mydas*	Threatened
Olive ridley	Olivacea kempii	Endangered
Leatherback sea turtle	*Dermochelys coriacea*	Endangered
Loggerhead sea turtle	Caretta caretta	Threatened
North Pacific Loggerhead DPS	Caretta caretta	Endangered
Chinook Salmon	*Oncorhynchus tshawytscha*	*Threatened/ Endangered*
Coho Salmon	*Oncorhynchus kisutch*	*Threatened/ Endangered*
Chum salmon	*Oncorhynchus keta*	*Threatened*
Sockeye salmon	*Oncorhynchus nerka*	*Threatened/ Endangered*
Steelhead Trout	*Oncorhynchus mykiss*	*Threatened/ Endangered*

Critical habitat for the North Pacific right whale was designated in the Bering Sea and in the Gulf of Alaska on April 8, 2008 (73 *Federal Register* 19000). Critical habitat was designated for the Eastern and Western DPSs of steller sea lions in 1993 (58 FR 45269). This critical habitat occurs along the Alaskan coastline and extends out 3,000 ft seaward in state and federally managed waters for the baseline or basepoint of each major rookery for the Eastern DPS and extends out 20 nm seaward in state and federally managed waters for the Western DPS.

Species Not Considered Further in this Consultation

Blue Whale

The blue whale has a cosmopolitan distribution and tends to be pelagic, only coming nearshore to feed and possibly to breed (Jefferson et al. 2008). Generally, blue whales are seasonal migrants between high latitudes in summer, where they feed, and low latitudes in winter, where they mate and give birth (Lockyer and Brown 1981). In the GOA, no detections of blue whales had been made since the late 1960s (NOAA 2004b; Calambokidis et al. 2009), until blue whale calls were recorded in the area during 1999–2002 (Stafford 2003; Stafford and Moore 2005; Moore et al. 2006; Stafford et al. 2007). Call types from both northeastern and northwestern

Pacific blue whales were recorded from July through December in the GOA, suggesting that two stocks use the area at that time (Stafford 2003; Stafford et al. 2007). Call rates peaked during August–November (Moore et al. 2006). In July 2004, three blue whales were sighted in the GOA, one on 14 July ~185 km southeast of PWS and two ~275 km southeast of PWS (NOAA 2004b; Calambokidis et al. 2009). These whales were thought to be part of the California feeding population (Calambokidis et al. 2009). Western blue whales are more likely to occur in the western portion of the GOA, southwest of Kodiak, where their calls have been detected (see Stafford 2003). Two blue whale sightings were also made in the Aleutians in August 2004 (Calambokidis et al. 2009). No blue whales were seen during surveys of the western GOA by Zerbini et al. (2006), or during surveys in the U.S. Navy training area east of Kodiak Island in April 2009 (Rone et al. 2010). Given the present rarity of these whales in the action area and the shut down procedures USGS will employ if a blue whale is observed regardless of distance from the *Langseth*, we conclude that the potential for this species to be exposed to the proposed survey is discountable; therefore blue whales are not likely to be adversely affected and we will not consider this species further in this Opinion.

Sei Whale

The distribution of the sei whale is not well known, but this whale is found in all oceans and appears to prefer mid-latitude temperate waters (Jefferson et al. 2008). Sei whales migrate from temperate zones occupied in winter to higher latitudes in the summer, where most feeding takes place (Gambell 1985a). During summer in the North Pacific, the sei whale can be found from the Bering Sea to the northern GOA and south to southern California, and in the western Pacific from Japan to Korea. Its winter distribution is concentrated at about 20°N, and sightings have been made between southern Baja California and the Islas Revilla Gigedo (Rice 1998). No breeding grounds have been identified for sei whales; however, calving is thought to occur from September to March. Moore et al. (2002b) made four sightings of six sei whales during summer surveys in the eastern Bering Sea, and one sighting south of the Alaska Peninsula between Kodiak and the Shumagin islands. No sei whales were seen during surveys of the GOA by Wade et al. (2003), Waite (2003), Zerbini et al. (2006), or Rone et al. (2010). Given the present rarity of these whales in the action area and the shut down procedures USGS will employ if a sei whale is observed regardless of distance from the *Langseth*, we conclude that the potential for this species to be exposed to the proposed survey is discountable; therefore sei whales are not likely to be adversely affected and we will not consider this species further in this Opinion.

North Pacific Right Whale

Historically, North Pacific right whales ranged across the entire North Pacific north of 35ºN and occasionally occurred as far south as 20ºN. North Pacific right whales summer in the northern North Pacific and Bering Sea, feeding off southern and western Alaska. The slope and abyssal plain in the western Gulf of Alaska was deemed an important are for right whale until the late 1960s (Shelden et al. 2005). In March 1979, a group of four right whales was seen in Yakutat Bay (Waite et al. 2003). However, there were no further reports of right whale sightings in the Gulf of Alaska until July 1998, when a single whale was seen southeast of Kodiak Island (Waite et al. 2003) and additional solitary animals were observed in the Barnabas Canyon in August 2004, 2005, and 2006 (NOAA unpublished data in Allen and Angliss 2010). One right whale was sighted in the Aleutian Islands south of Umiak Pass in September 2004 (Wade et al. 2010).

Right whale acoustic detections were made south of the Alaska Peninsula and to the east of Kodiak Island in 2000 during August and September (see Waite et al. 2003; Mellinger et al. 2004b), but no acoustic detections were made from April to August 2003 (Munger et al. 2008) or in April 2009 (Rone et al. 2010).

Given the low population size of these whales, the present rarity of the whales in the action area and the shut down procedures USGS will employ if a North Pacific right whale is observed regardless of distance from the *Langseth*, we conclude that the potential for this species to be exposed to the proposed survey is discountable; therefore, the North Pacific right whale is not likely to be adversely affected and we will not consider this species further in this Opinion.

North Pacific Right Whale Critical Habitat

In 2008 NMFS designated two areas as critical habitat for the North Pacific right whale: one area in the western Gulf of Alaska and one in the southeast Bering Sea. The primary constituent elements deemed necessary for the conservation of North Pacific right whales include copepods (Calanus marshallae, Neocalanus cristatus, and N. plumchris), and the euphausiids (Thysanoessa Raschii).

The critical habitat in the Bering Sea would not be exposed to acoustic signals from the airgun as the Aleutian Islands should form an effective barrier to these signals. The critical habitat in the Gulf of Alaska is located south of Kodiak Island, however, none of the proposed transect lines enter the critical habitat and the survey will occur far enough away from the critical habitat area that received sound levels within the habitat will not exceed 160 dB re 1 μPa(rms.). For these reasons, we do not expect critical habitat to be adversely affected by acoustic signals from the proposed survey, therefore, we will not consider critical habitat further in this Opinion.

Steller Sea Lion Critical Habitat

Designated critical habitat in the Gulf of Alaska occurs along the Alaskan coastline and extends out 3, 000 ft seaward in state and federally managed waters from the baseline or basepoint of each major rookery for the Eastern DPS and extends out 20 nm seaward in state and federally managed waters for the Western DPS. Essential features of critical habitat that are essential for the conservation of steller sea lions include foraging habitat. None of the proposed transect lines enter steller sea lion critical habitat as the nearest the survey will occur to the Alanskan coastline is 108 nm. The survey will occur far enough away from the critical habitat area that received sound levels, if any, within critical habitat are not expected to diminish the conservation value of critical habitat through changes in quantity, quality or distribution of steller sea lion prey species, therefore, we will not consider critical habitat further in this Opinion.

Green, Olive ridley, and Loggerhead Sea Turtles

Only four species of sea turtles have been observed in Alaska waters between 1960 and 1998: the green, Olive ridley,leatherback and loggerhead sea turtles (ADFG 2011; Hodge and Wing 2000). The ocean waters of the Gulf of Alaska have an average sea surface temperature in summer in the upper 100 m (328 ft) of approximately 51.8 degree Fahrenheit (°F) (11 degrees Celsius [°C]). Most hard-shell turtles seek optimal seawater temperatures near 65°F and are cold-stressed at seawater temperatures below 50°F (Davenport 1997). At temperatures below 15°C, green and ridley sea turtles become semidormant and hardly move (Milton and Lutz 2003). Loggerhead

sea turtles exposed to excessive low temperatures have experienced abrupt failure in pH homeostasis and a sharp increase in blood lactate levels (Milton and Lutz 2003). At 10°C loggerhead sea turtles were lethargic and "floated" (Milton and Lutz 2003).

In Alaska, only 9 green sea turtle occurrences, 2 olive ridley occurrences, and 2 loggerheads were documented between 1960 and 2006 (Hodge and Wing 2000; Navy 2006). Therefore, although sightings of these sea turtle species have been documented in Alaska, most of these involve individuals that were either cold-stressed, likely to become cold-stressed, or already deceased (Hodge and Wing 2000; McAlpine et al. 2002). Thus, the Gulf of Alaska is considered to be outside the normal range for these sea turtle species and will not be considered further in this Opinion.

Pacific Salmon and Trout

Juvenile and maturing Chinook, coho, chum, sockeye salmon and steelhead trout occur with the Gulf of Alaska's coastal and marine waters. These salmonids migrate from the west coast of the United States to forage before returning to their natal streams to spawn. Salmonid occurrence is fairly close to the Alaskan shoreline as listed below:

Chinook salmon

Early life history stages for chinook occur in freshwater but juveniles and adults utilize marine habitats within the Gulf of Alaska. The majority of marine juveniles are found within 28 km of the coast (PFMC 2000). They tend to concentrate around areas of pronounced coastal upwelling.

Coho salmon

Coho salmon from Oregon have been taken in offshore waters near Kodiak Island in the northern Gulf of Alaska. In strong upwelling years, coho salmon are more dispersed offshore, whereas in weak upwelling years they concentrate near submarine canyons and areas of consistent upwelling. Offshore, juvenile coho are generally found in waters over the continental shelf, ranging from 23 to 46 mi (37 to 74 km) from shore.

Chum salmon

Juveniles and adult chum utilize marine habitats within the Gulf of Alaska. Juvenile chum occur along the coast of North America and Alaska in a band that extends out to 19 nm (36 km) (Salo 1991). Juvenile chum migrations follow the Gulf of Alaska coastal belt to the north, west, and south during their first summer at sea (Salo 1991). While overall migrations patterns of juvenile chum salmon within the Gulf of Alaska are understood, nearshore residency times and offshore migrations patterns are still unclear (Salo 1991). Maturing fish destined for North American streams are widely distributed throughout the Gulf of Alaska during the spring and summer (Salo 1991).

Sockeye salmon

Juveniles and adult sockeye utilize marine habitats within the Gulf of Alaska and vicinity. Soon after entering the ocean, juvenile sockeye salmon (excluding those from Bristol Bay) begin moving north into the Gulf of Alaska where they remain along the coastal belt until late-fall or early-winter. They then disperse offshore moving west and south (Emmett et al. 1991). In the Gulf of Alaska sockeye move north during the spring and summer then south and west during the

winter (Emmett et al. 1991). The Snake river ESU has remained at very low levels of only a few hundred fish.

Steelhead trout
Steelhead trout are found from central California to the Bering Sea and Bristol Bay coastal streams of Alaska. Juvenile and adult steelhead utilize the marine environment in the Gulf of Alaska and vicinity.

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

The lateral line system of a fish allows for sensitivity to sound (Hastings and Popper 2005). This system is a series of receptors along the body of the fish that detects water motion relative to the fish that arise from sources within a few body lengths of the animal. The sensitivity of the lateral line system is generally from below 1 Hz to a few hundred Hz (Coombs and Montgomery 1999; Popper and Schilt 2009).

While studies on the effect of sound on the lateral line are limited, the work of Hasting et al.(1996) showing limited sensitivity to within a few body lengths and to sounds below a few hundred Hz . Broadly, fish can be categorized as either hearing specialists or hearing generalists (Scholik and Yan 2002). Of the fish species with distributions overlapping the Gulf of Alaska for which hearing sensitivities are known, most are hearing generalists, including salmonid species. Fish in the hearing generalists category are limited to detection of the particle motion component of low-frequency sounds at relatively high sound intensities (Amoser and Ladich 2005). The hearing capability of Atlantic salmon (*Samo salar*), a hearing generalist, indicates a rather low sensitivity to sound (Hawkins and Johnstone 1978). Laboratory experiments yielded responses only to 0.58 kHz and only at high sound levels. The salmon's poor hearing is likely due to the lack of a link between the swim bladder and inner ear (Jørgensen et al. 2004).

Based on the above salmon and trout are expected to mostly occur in the coastal and nearshore marine areas within the Gulf of Alaska. The proposed survey would begin approximately 108 nm from the Alaskan shoreline and continue to survey out to approximately 351 nm from the shoreline. Given the distances of the proposed survey from the areas where listed salmonids are expected to occur (the particle motion associated with the seismic signals to which salmonids are sensitive to would attenuate relatively quickly) and the sensitivity of the lateral line (limited sensitivity to within a few body lengths) to acoustic sounds we would not expect listed salmonids

to experience behavioral disruptions that arise to the level of harassment, if they experience any disruptions at all. If listed salmonids are exposed to acoustic signals in the deeper waters of the Gulf of Alaska closer to the proposed survey then we would only expect temporary behavioral disruptions, not arising to the level of harassment because of the sensitivity of the lateral line, the expected avoidance reaction of listed salmonids to harmful levels of acoustic signals and the movement of the survey vessel along the survey trackline. Based on the above we will not consider listed salmon and trout further in this Opinion.

Status of Species Considered in this Biological Opinion

The remainder of this section consists of narratives for each of the threatened and endangered species that occur in the action area and that may be adversely affected by the oroposed survey in the Gulf of Alaska. Each narrative presents a summary of information on the distribution and population structure of each species to provide a foundation for the exposure analyses that appear later in this Opinion. A summary of information on the threats to the species and the species' status given those threats is provided as points of reference for the subsequent jeopardy determinations. That is, NMFS relies on a species' status and trend to determine whether or not an action's direct or indirect effects are likely to increase the species' probability of becoming extinct.

More detailed background information on the status of these species and critical habitat can be found in a number of published documents including status reviews, recovery plans for the blue whale (NMFS 1998b), fin whales (NMFS 2010b), fin and sei whale (NMFS 1998a), humpback whale (NMFS 1991), right whale (NMFS 2004), sperm whale (NMFS 2010c), a status report on large whales prepared by Perry et al. (1999a) and the status review and recovery plan for the leatherback sea turtle (NMFS and USFWS 1998; NMFS and USFWS 2007).

Fin whale

Distribution

Fin whales are distributed widely in every ocean except the Arctic Ocean. In the North Pacific Ocean, fin whales occur in summer foraging areas in the Chukchi Sea, the Sea of Okhotsk, around the Aleutian Islands, and the Gulf of Alaska; in the eastern Pacific, they occur south to California; in the western Pacific, they occur south to Japan. Fin whales in the eastern Pacific winter from California south; in the western Pacific, they winter from the Sea of Japan, the East China and Yellow Seas, and the Philippine Sea (Gambell 1985).

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 Meyers, 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 1985).

In the Southern Hemisphere, fin whales are distributed broadly south of 50° S in the summer and migrate into the Atlantic, Indian, and Pacific Oceans in the winter, along the coast of South

America (as far north as Peru and Brazil), Africa, and the islands in Oceania north of Australia and New Zealand (Gambell 1985).

Fin whales are common off the Atlantic coast of the United States in waters immediately off the coast seaward to the continental shelf (about the 1,000-fathom contour). In this region, they tend to occur north of Cape Hatteras where they accounted for about 46 percent of the large whales observed in surveys conducted between 1978 and 1982. During the summer months, fin whales in this region tend to congregate in feeding areas between 41°20'N and 51°00'N, from shore seaward to the 1,000-fathom contour.

In the Atlantic Ocean, Clark (1995) reported a general southward pattern of fin whale migration in the fall from the Labrador and Newfoundland region, south past Bermuda, and into the West Indies. The overall distribution may be based on prey availability, and fin whales are found throughout the action area for this consultation in most months of the year. This species preys opportunistically on both invertebrates and fish (Watkins et al. 1984). They feed by filtering large volumes of water for the associated prey. Fin whales are larger and faster than humpback and right whales and are less concentrated in nearshore environments.

Population Structure

Fin whales have two recognized subspecies: Balaoptera physalus physalus (Linnaeus 1758) occurs in the North Atlantic Ocean while B. p. quoyi (Fischer 1829) occurs in the Southern Ocean. Globally, fin whales are sub-divided into three major groups: Atlantic, Pacific, and Antarctic. Within these major areas, different organizations use different population structure.

In the North Atlantic Ocean, the IWC recognizes seven management units or “stocks” of fin whales: (1) Nova Scotia, (2) Newfoundland-Labrador, (3) West Greenland, (4) East Greenland-Iceland, (5) North Norway, (6) West Norway-Faroe Islands, and (7) British Isles-Spain-Portugal. In addition, the population of fin whales that resides in the Ligurian Sea, in the northwestern Mediterranean Sea is believed to be genetically distinct from other fin whales populations (as used in this Opinion, “populations” are isolated demographically, meaning, they are driven more by internal dynamics — birth and death processes — than by the geographic redistribution of individuals through immigration or emigration. Some usages of the term “stock” are synonymous with this definition of “population” while other usages of “stock” are not).

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. (1984) concluded that there were five possible “stocks” of fin whales within the North Pacific based on histological analyses and tagging experiments: (1) East and West Pacific that intermingle around the Aleutian Islands; (2) East China Sea; (3) British Columbia; (4) Southern-Central California to Gulf of Alaska; and (5) Gulf of California. Based on genetic analyses, Berube et al. (1998) concluded that fin whales in the Sea of Cortez represent an isolated population that has very little genetic exchange with other populations in the North Pacific Ocean (although the geographic distribution of this population and other populations can overlap seasonally). They also concluded that fin whales in the Gulf of St. Lawrence and Gulf of Maine are distinct from fin whales found off Spain and in the Mediterranean Sea.

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

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

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

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

In its draft recovery plan for fin whales, NMFS recognized three populations in U.S. Pacific waters: Alaska (Northeast Pacific), California/Oregon/Washington, and Hawai'i (Barlow et al. 1997; Hill et al. 1997). We assume that individuals from the Alaska "population" of fin whales are the whales that would be exposed to the activities considered in this consultation.

Threats to the Species

Natural threats. Natural sources and rates of mortality are largely unknown, but Aguilar and Lockyer (1987) suggest annual natural mortality rates may range from 0.04 to 0.06. Although these results are based on studies of fin whales in the northeast Atlantic, there are no comparable estimates for fin whales in the Pacific Ocean. The occurrence of the nematode Crassicauda boopis appears to increase the potential for kidney failure in fin whales and may be preventing some fin whale stocks from recovering from whaling (Lambertsen 1992, as cited in Perry et al. 1999). Killer whale or shark attacks may injure or kill very young or sick whales (Perry et al. 1999, Tomilin 1967).

Anthropogenic threats. Three human activities are known to threaten fin whales: whaling, commercial fishing, and shipping. Historically, whaling represented the greatest threat to every population of fin whales and was ultimately responsible for listing fin whales as an endangered species. As early as the mid-seventeenth century, the Japanese were capturing fin, blue (Balaenoptera musculus), and other large whales using a fairly primitive open-water netting technique (Tønnessen and Johnsen 1982, Cherfas 1989). In 1864, explosive harpoons and steam-powered catcher boats were introduced in Norway, allowing the large-scale exploitation of previously unobtainable whale species. After blue whales were depleted in most areas, fin

whales became the focus of whaling operations and more than 700,000 fin whales were landed in the Southern Hemisphere alone between 1904 and 1979 (IWC 1995).

As its legacy, whaling has reduced fin whales to a fraction of their historic population size and, as a result, makes it easier for other human activities to push fin whales closer to extinction. Otherwise, whaling currently does not threaten every fin whale population, although it may threaten specific populations.

From 1904 to 1975, the IWC estimates that 703,693 fin whales captured and killed in Antarctic whaling operations (IWC 1990). Whaling in the Southern Oceans originally targeted humpback whales, but by 1913, those whales had became rare so whalers shifted their focus to fin and blue whales (Mizroch et al. 1984b). From 1911 to 1924, whalers killed 2,000–5,000 fin whales each year. After the introduction of factory whaling ships in 1925, the number of whales killed each year increased substantially: from 1931 to 1972, whalers killer about 511,574 fin whales (Kawamura 1994). In 1937 alone, whalers are reported to have killed more than 28,000 fin whales. From 1953 to 1961, the number of fin whales killed each year averaged around 25,000. In 1962, whalers appeared to shift their focus to sei whale as fin whales became scarce. By 1974, whalers killed fewer than 1,000 fin whales.

Recently released Soviet whaling records indicate a discrepancy between reported and actual fin whale catch numbers by whalers from the former USSR in southern waters between 1947 and 1980 (Zemsky et al. 1995). The former USSR previously reported 52,931 whales caught; however, the data that was released recently suggests that only 41,984 were killed.

In the Antarctic Ocean, fin whales were hunted by Japanese whalers who have been allowed to kill up to 10 fin whales each year for the 2005-2006 and 2006-2007 seasons under an Antarctic Special Permit. The Japanese whalers plan to kill 50 fin whales per year starting in the 2007-2008 season and continuing for the next 12 years.

Fin whales are also hunted in subsistence fisheries off West Greenland. In 2004, 5 males and 6 females were killed and landed; 2 other fin whales were struck and lost in the same year. In 2003 2 males and 4 females were landed and 2 other fin whales were struck and lost (IWC 2005). Between 2003 and 2007, the IWC set a catch limit of up to 19 fin whales in this subsistence fishery (IWC 2005), however, the IWC's Scientific Committee recommended limiting the number of fin whales killed in this fishery to 1 to 4 individuals until accurate population estimates are produced.

Despite anecdotal observations from fishermen which suggest that large whales swim through their nets rather than get caught in them (NMFS 2000), fin whales have been entangled by fishing gear off Newfoundland and Labrador in small numbers: a total of 14 fin whales are reported to have been captured in coastal fisheries in those two provinces between 1969 and 1990 (Lien 1994, Perkins and Beamish 1979). Of these 14 fin whales, 7 are known to have died as a result of that capture, although most of the animals that died were less than 15 meters in length (Lien 1994). Between 1999 and 2005, there were 10 confirmed reports of fin whales being entangled in fishing gear 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, Fin whales were injured in 1 of

the entanglements and killed in 3 entanglements. These data suggest that, despite their size and strength, fin whales are likely to be entangled and, in some cases, killed by gear used in modern fisheries.

Fin whales are also killed and injured in collisions with vessels more frequently than any other whale. Of 92 fin whales that stranded along the Atlantic Coast of the U.S. between 1975 and 1996, 31 (33%) showed evidence of collisions with ships (Laist et al. 2001). Between 1999 and 2005, there were 15 reports of fin 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 which were reported as having resulted in the death of 11 fin whales.

Ship strikes were identified as a known or potential cause of death in 8 (20%) of 39 fin whales that stranded on the coast of Italy in the Mediterranean Sea between 1986 and 1997 (Laist et al. 2001). Throughout the Mediterranean Sea, 46 of the 287 fin whales that are recorded to have stranded between 1897 and 2001 were confirmed to died from injuries sustained by ship strikes (Panigada et al. 2006). Most of these fin whales (n = 43) were killed between 1972 and 2001 and the highest percentage (37 of 45 or ~82%) were killed in the Ligurian Sea and adjacent waters, where the Pelagos Sanctuary for Marine Mammals was established. In addition to these ship strikes, there are numerous reports of fin whales being injured as a result of ship strikes off the Atlantic coast of France and the United Kingdom (Jensen and Silber 2003).

Status

Fin whales were listed as endangered under the ESA in 1970. In 1976, the IWC protected fin whales from commercial whaling (Allen 1980). Fin whales are listed as endangered on the IUCN Red List of Threatened Animals (Baillie and Groombridge 1996). They are also protected by the Convention on International Trade in Endangered Species of wild flora and fauna and the MMPA. Critical habitat has not been designated for fin whales.

It is difficult to assess the current status of fin whales because (1) there is no general agreement on the size of the fin whale population prior to whaling and (2) estimates of the current size of the different fin whale populations vary widely (NMFS 2007). We may never know the size of the fin whale population prior to whaling. The most current estimate of the population size of fin whales in the Pacific Ocean is 85,200 (no coefficient of variance or confidence interval was provided) based on the history of catches and trends in catches per unit of effort (IWC 1979). Based on surveys conducted south of 30°S latitude between 1978 and 1988, fin whales in the Southern Ocean were estimated to number about 400,000 (IWC 1979; no coefficient of variance or confidence interval was provided).

Chapman (1976) estimated the “original” population size of fin whales off Nova Scotia as 1,200 and 2,400 off Newfoundland, although he offered no explanation or reasoning to support that estimate. Sergeant (1977) suggested that between 30,000 and 50,000 fin whales once populated the North Atlantic Ocean based on assumptions about catch levels during the whaling period. Sigurjónsson (1995) estimated that between 50,000 and 100,000 fin whales once populated the North Atlantic, although he provided no data or evidence to support that estimate. More recently, Palumbi and Roman (2006) estimated that about 360,000 fin whales (95% confidence interval =

249,000 - 481,000) populated the North Atlantic Ocean before whaling based on mutation rates and estimates of genetic diversity.

Similarly, estimates of the current size of the different fin whale populations and estimates of their global abundance also vary widely. The draft recovery plan for fin whales accepts a minimum population estimate of 2,362 fin whales for the North Atlantic Ocean (NMFS 2007); however, the recovery plan also states that this estimate, which is based on on shipboard and aerial surveys conducted in the Georges Bank and Gulf of St. Lawrence in 1999 is the "best" estimate of the size of this fin whale population (NMFS 2006, 2007). However, based on data produced by surveys conducted between 1978-1982 and other data gathered between 1966 and 1989, Hain et al. (1992) estimated that the population of fin whales in the western North Atlantic Ocean (specifically, between Cape Hatteras, North Carolina, and Nova Scotia) numbered about 1,500 whales in the winter and 5,000 whales in the spring and summer. Because authors do not always reconcile "new" estimates with earlier estimates, it is not clear whether the current "best" estimate represents a refinement of the estimate that was based on older data or whether the fin whale population in the North Atlantic has declined by about 50% since the early 1980s.
The East Greenland-Iceland fin whale population was estimated at 10,000 animals (95 % confidence interval = 7,600 - 14,200), based on surveys conducted in 1987 and 1989 (Buckland et al. 1992). The number of eastern Atlantic fin whales, which includes the British Isles-Spain-Portugal population, has been estimated at 17,000 animals (95% confidence interval = 10,400 - 28,900; Buckland et al. 1992). These estimates are both more than 15 years old and the data available do not allow us to determine if they remain valid.

Forcada et al. (1996) estimated there were 3,583 fin whales in the western Mediterranean (standard error = 967; 95% confidence interval = 2,130 - 6,027), which is similar to an estimate published by Notarbartolo-di-Sciara et al. (2003). In the Mediterraneans' Ligurian Sea (which includes the Pelagos Whale Sanctuary and the Gulf of Lions), Forcada et al. (1995) estimated there were 901 fin whales (standard error = 196.1).

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

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

Diving and Social Behavior

The percentage of time fin whales spend at the surface varies. Some authors have reported that fin whales make 5-20 shallow dives with each of these dives lasting 13-20 seconds followed by a deep dive lasting between 1.5 and 15 minutes (Gambell 1985). Other authors have reported that the fin whale's most common dives last between 2 and 6 minutes, with 2 to 8 blows between dives (Hain et al. 1992, Watkins 1981).

In waters off the Atlantic Coast of the U.S. individual fin whales or pairs represented about 75% of the fin whales observed during the Cetacean and Turtle Assessment Program (Hain et al. 1992). Individual whales or groups of less than five individuals represented about 90% of the observations (out of 2,065 observations of fin whales, the mean group size was 2.9, the modal value was 1, and the range was 1 – 65 individuals; Hain et al. 1992).

Vocalizations and Hearing

The sounds fin whales produce underwater are one of the most studied Balaenoptera sounds. Fin whales produce a variety of low-frequency sounds in the 10-200 Hz band (Watkins 1981; Watkins et al. 1987a; Edds 1988; Thompson et al. 1992). The most typical signals are long, patterned sequences of short duration (0.5-2s) infrasonic pulses in the 18-35 Hz range (Patterson and Hamilton 1964). Estimated source levels are as high as 190 dB (Patterson and Hamilton 1964; Watkins et al. 1987a; Thompson et al. 1992; McDonald et al. 1995). In temperate waters intense bouts of long patterned sounds are very common from fall through spring, but also occur to a lesser extent during the summer in high latitude feeding areas (Clark and Charif 1998). Short sequences of rapid pulses in the 20-70 Hz band are associated with animals in social groups (McDonald et al. 1995, Clark personal communication, McDonald personal communication). Each pulse lasts on the order of one second and contains twenty cycles (Tyack 1999).

During the breeding season, fin whales produce a series of pulses in a regularly repeating pattern. These bouts of pulsing may last for longer than one day (Tyack 1999). The seasonality and stereotype of the bouts of patterned sounds suggest that these sounds are male reproductive displays (Watkins et al. 1987a), while the individual counter-calling data of McDonald et al. (1995) suggest that the more variable calls are contact calls. Some authors feel there are geographic differences in the frequency, duration and repetition of the pulses (Thompson et al. 1992).

As with other vocalizations produced by baleen whales, the function of fin whale vocalizations is unknown, although there are numerous hypotheses (which include: maintenance of inter-individual distance, species and individual recognition, contextual information transmission, maintenance of social organization, location of topographic features, and location of prey resources; see the review by Thompson et al. 1992 for more information on these hypotheses). 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 (Payne and Webb 1971; Edds-Walton 1997). 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).

Cetaceans have an auditory anatomy that follows the basic mammalian pattern, with some modifications to adapt to the demands of hearing in the sea. The typical mammalian ear is divided into the outer ear, middle ear, and inner ear. The outer ear is separated from the inner ear by the tympanic membrane, or eardrum. In terrestrial mammals, the outer ear, eardrum, and middle ear function to transmit airborne sound to the inner ear, where the sound is detected in a fluid. Since cetaceans already live in a fluid medium, they do not require this matching, and thus do not have an air-filled external ear canal. The inner ear is where sound energy is converted into neural signals that are transmitted to the central nervous system via the auditory nerve. Acoustic energy causes the basilar membrane in the cochlea to vibrate. Sensory cells at different positions along the basilar membrane are excited by different frequencies of sound (Tyack 1999). Baleen whales have inner ears that appear to be specialized for low-frequency hearing. In a study of the morphology of the mysticete auditory apparatus, Ketten (1997) hypothesized that large mysticetes have acute infrasonic hearing[3].

Humpback Whale

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 reproduce and give birth to calves) and cooler, temperate or sub-Arctic waters in summer months (where they feed). In their summer foraging areas and winter calving areas, humpback whales tend to occupy shallower, coastal waters; during their seasonal migrations, however, humpback whales disperse widely in deep, pelagic waters and tend to avoid shallower coastal waters (Winn and Reichley 1985).

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

In the Atlantic Ocean, humpback whales range from the mid-Atlantic bight, the Gulf of Maine, across the southern coast of Greenland and Iceland, and along coast of Norway in the Barents Sea. These humpback whales migrate to the western coast of Africa and the Caribbean Sea during the winter.

In the Southern Ocean, humpback whales occur in waters off Antarctica. These whales migrate to the waters off Venezuela, Brazil, southern Africa, western and eastern Australia, New Zealand, and islands in the southwest Pacific during the austral winter. A separate population of humpback whales appears to reside in the Arabian Sea in the Indian Ocean off the coasts of Oman, Pakistan, and India (Mikhalev 1997).

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3 Infrasonic hearing is the capability of hearing sounds too low for humans to hear (i.e., below 20 Hz).

Population Structure

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

NMFS' Stock Assessment Reports recognize four "stocks" of humpback whales in the North Pacific Ocean, based on genetic and photo-identification studies: two Eastern North Pacific stocks, one Central North Pacific stock, and one Western Pacific stock (Hill and DeMaster 1998). The first two of these "stocks" are based on where these humpback whales winter: the central North Pacific "stock" winters in the waters around Hawai'i while the eastern North Pacific "stock" (also called the California-Oregon-Washington-Mexico stock) winters along coasts of Central America and Mexico. However, Calambokidis et al. (1997) identified humpback whales from Southeast Alaska (central North Pacific), the California-Oregon-Washington (eastern North Pacific), and Ogasawara Islands (Japan, Western Pacific) groups in the Hawai'ian Islands during the winter; humpback whales from the Kodiak Island, Southeast Alaska, and British Columbia groups in the Ogasawara Islands; and whales from the British Columbia, Southeast Alaska, Prince William Sound, and Shumagin-Aleutian Islands groups in Mexico.

Herman (1979), however, presented extensive evidence and various lines of reasoning to conclude that the humpback whales associated with the main Hawai'ian Islands immigrated to those waters only in the past 200 years. Winn and Reichley (1985) identified genetic exchange between the humpback whales that winter off Hawai'i and those that winter off Mexico (with further mixing on feeding areas in Alaska) and suggested that the humpback whales that winter in Hawai'i may have emigrated from wintering areas in Mexico. Based on these patterns of movement, we conclude that the various "stocks" of humpback whales are not true populations or, at least, they represent populations that experience substantial levels of immigration and emigration.

A "population" of humpback whales winters in an area extending from the South China Sea east through the Philippines, Ryukyu Retto, Ogasawara Gunto, Mariana Islands, and Marshall Islands (Rice 1998). Based on whaling records, humpback whales wintering in this area have also occurred in the southern Marianas through the month of May (Eldredge 1991). There are several recent records of humpback whales in the Mariana Islands, at Guam, Rota, and Saipan during January through March (Darling and Mori 1993; Eldredge 1991, 2003; Taitano 1991). During the 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 2007, Calambokidis 1997, 2001).

Between 2004 and 2006, an international group of whale researchers coordinated their surveys to conduct a comprehensive assessment of the population structure, levels of abundance, and status of humpback whales in the North Pacific (Calambokidis et al. 2008). That effort identified a total of 7,971 unique individuals from photographs taken during close approaches.

In the Atlantic Ocean, humpback whales aggregate in four feeding areas in the summer months: (1) Gulf of Maine, eastern Canada, (2) west Greenland, (3) Iceland and (4) Norway (Katona and Beard 1990, Smith et al. 1999). The principal breeding range for these whales lies from the Antilles and northern Venezuela to Cuba (Winn et al. 1975, Balcomb and Nichols 1982, Whitehead and Moore 1982). The largest contemporary breeding aggregations occur off the Greater Antilles where humpback whales from all of the North Atlantic feeding areas have been identified from photographs (Katona and Beard 1990, Clapham et al. 1993b, Mattila et al. 1994, Palsbøll et al. 1997, Smith et al. 1999, Stevick et al. 2003a). Historically, an important breeding aggregation was located in the eastern Caribbean based on the important humpback whale fisheries this region supported (Mitchell and Reeves 1983, Reeves et al. 2001, Smith and Reeves 2003). Although sightings persist in those areas, modern humpback whale abundance appears to be low (Winn et al. 1975, Levenson and Leapley 1978, Swartz et al. 2003). Winter aggregations also occur at the Cape Verde Islands in the Eastern North Atlantic (Reiner et al. 1996, Reeves et al. 2002, Moore et al. 2003). In another example of the “open” structure of humpback whale populations, an individual humpback whale migrated from the Indian Ocean to the South Atlantic Ocean and demonstrated that individual whales may migrate from one ocean basin to another (Pomilla and Rosenbaum 2005).

As discussed previously, a separate population of humpback whales appears to reside in the Arabian Sea in the Indian Ocean off the coasts of Oman, Pakistan, and India (Mikhalev 1997).

Threats to the Species

Natural threats. There is limited information on natural phenomena that kill or injure humpback whales. We know that humpback whales are killed by orcas (Dolphin 1989, Florez-González et al. 1984, Whitehead and Glass 1985) and are probably killed by false killer whales and sharks. Because 7 female and 7 male humpback whales stranded on the beaches of Cape Cod and had died from toxin produced by dinoflagellates between November 1987 and January 1988, we also know that adult and juvenile humpback whales are killed by naturally-produced biotoxins (Geraci et al. 1989).

Other natural sources of mortality, however, remain largely unknown. Similarly, we do not know whether and to what degree natural mortality limits or restricts patterns of growth or variability in humpback whale populations.

Anthropogenic threats. Three human activities are known to threaten humpback whales: whaling, commercial fishing, and shipping. Historically, whaling represented the greatest threat to every population of humpback whales and was ultimately responsible for listing humpback whales as an endangered species. From 1900 to 1965, nearly 30,000 whales were taken in modern whaling operations of the Pacific Ocean. Prior to that, an unknown number of humpback whales were taken (Perry et al. 1999). In 1965, the IWC banned commercial hunting of humpback whales in the Pacific Ocean. As its legacy, whaling has reduced humpback whales to a fraction of their

historic population size and, as a result, makes it easier for other human activities to push these whales closer to extinction.

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 are reported to have been captured in coastal fisheries in those two provinces between 1969 and 1990 (Lien 1994, Perkins and Beamish 1979). Of these whales, 94 are known to have died as a result of that capture, although, like fin whales, most of the animals that died were smaller: less than 12 meters in length (Lien 1994). These data suggest that, despite their size and strength, humpback whales are likely to be entangled and, in some cases, killed by gear used in modern fisheries.

There are also reports of entangled humpback whales from the Hawai'ian 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. Also in 1996, a vessel from Pacific Missile Range Facility in Hawai'i rescued an entangled humpback, removing two crab pot floats from the whale. From 2001 through 2006, there were 23 reports of entangled humpback whales in Hawai'ian waters; 16 of these reports were from 2005 and 2006.

Many of the entangled humpback whales observed in Hawai'ian waters brought the gear with them from higher latitude feeding grounds; for example, the whale the U.S. Navy rescued in 1996 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 Hawai'ian Islands have been confirmed to have been entangled in gear from Alaska. Nevertheless, humpback whales are also entangled in fishing gear in the Hawai'ian Islands. Since 2001, there have been 5 observed interactions between humpback whales and gear associated with the Hawai'i-based longline fisheries (NMFS 2008). 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.

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 reports, 95 entanglements were confirmed resulting in the injury of 11 humpback whales and the death of 9 whales. No information is available on the number of humpback whales that have been killed or seriously injured by interactions with fishing fleets outside of U.S. waters.

The number of humpback whales killed by ship strikes is exceeded only by fin whales (Jensen and Silber 2003). On the Pacific coast, a humpback whale is killed about every other year by ship strikes (Barlow et al. 1997). 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). Of 123 humpback whales that stranded along the Atlantic Coast of the U.S. between 1975 and 1996, 10 (8.1%) showed evidence of collisions with ships (Laist et al. 2001). Between 1999 and 2005, there were 18 reports of

humpback whales being struck by vessels along the Atlantic Coast of the U.S. and the Maritime Provinces of Canada (Cole et al. 2005, Nelson et al. 2007). Of these reports, 13 were confirmed as ship strikes which were reported as having resulted in the death of 7 humpback whales. Despite several literature searches, we did not identify information on the number of humpback whales killed or seriously injured by ship strikes outside of U.S. waters.

In addition to ship strikes in North America and Hawai'i, there are several reports of humpback whales being injured as result of ship strikes off the Antarctic Peninsula, in the Caribbean Sea, the Mediterranean Sea, off Australia, Bay of Bengal (Indian Ocean), Brazil, New Zealand, Peru, and South Africa.

Status

Humpback whales were listed as endangered under the ESA in 1973. Humpback whales are listed as endangered on the IUCN Red List of Threatened Animals (Baillie and Groombridge 1996). They are also protected by the Convention on International Trade in Endangered Species of wild flora and fauna and the MMPA. Critical habitat has not been designated for humpback whales.

It is difficult to assess the current status of humpback whales for the same reasons that it is difficult to assess the status of fin whales: (1) there is no general agreement on the size of the humpback whale population prior to whaling and (2) estimates of the current size of the different humpback whale populations vary widely and produce estimates that are not always comparable to one another, although robust estimates of humpback whale populations in the western North Atlantic have been published. We may never know the size of the humpback whale population prior to whaling.

Winn and Reichley (1985) argued that the global population of humpback whales consisted of at least 150,000 whales in the early 1900s, with the largest population historically occurring in the Southern Ocean. Based on analyses of mutation rates and estimates of genetic diversity, Palumbi and Roman (2006) concluded that there may have been as many as 240,000 (95% confidence interval = 156,000 – 401,000) humpback whales in the North Atlantic before whaling began. In the western North Atlantic between Davis Strait, Iceland and the West Indies, Mitchell and Reeves (1983) estimated there were at least 4,685 humpback whales in 1865 based on available whaling records (although the authors note that this does not represent a "pre-exploitation estimate" because whalers from Greenland, the Gulf of St. Lawrence, New England, and the Caribbean Sea had been hunting humpback whales before 1865).

Estimates of the number of humpback whales occurring in the different populations that inhabit the Northern Pacific population have risen over time. In the 1980s, the size of the North Pacific humpback whale population was estimated to range from 1,407 to 2,100 (Baker 1985; Darling and Morowitz 1986; Baker and Herman 1987). By the mid-1990s, the population was estimated to consist of about 6,000 whales (standard error = 474) in the North Pacific (Calambokidis et al. 1997; Cerchio 1998; Mobley et al. 1999).

As discussed previously, between 2004 and 2006, an international group of whale researchers coordinated their surveys to conduct a comprehensive assessment of the population structure,

levels of abundance, and status of humpback whales in the North Pacific (Calambokidis et al. 2008). That effort identified a total of 7,971 unique individuals from photographs taken during close approaches. Of this total,, 4,516 individuals were identified at wintering regions in at least one of the three seasons in which the study surveyed wintering area and 4,328 individuals were identified at least once at feeding areas in one of the two years in which the study surveyed feeding areas. Based on the results of that effort, Calambokidis et al. (2008) estimated that the current population of humpback whales in the North Pacific Ocean consisted of about 18,300 whales, not counting calves. Almost half of the humpback whales that were estimated to occur in wintering areas, or about 8,000 humpback whales, occupy the Hawai'ian Islands during the winter months.

In the North Atlantic, Stevick et al. (2003) estimated the size of the humpback whale population between 1979 and 1993 by applying statistical analyses that are commonly used in capture-recapture studies to individual humpback whales that were identified based on natural markings. Between 1979 and 1993, they estimated that the North Atlantic populations (what they call the "West Indies breeding population") consisted of between 5,930 and 12,580 individual whales. The best estimate they produced (11,570; 95% confidence interval = 10,290 -13,390) was based on samples from 1992 and 1993. If we assume that this population has grown according to the instantaneous rate of increase Stevick et al. (2003) estimated for this population (r = 0.0311), this would lead us to estimate that this population might consist of about 18,400 individual whales in 2007-2008.

Regardless of which of these estimates, if any, most closely correspond to the actual size and trend of the humpback whale population, all of these estimates suggest that the global population of humpback whales consists of tens of thousands of individuals, that the North Atlantic population consists of at least 2,000 individuals and the North Pacific population consists of about 18,000 individuals. Based on ecological theory and demographic patterns derived from several hundred imperiled species and populations, humpback 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 humpback whales will have elevated extinction probabilities because of exogenous threats caused by anthropogenic activities (primarily whaling, entanglement, and ship strikes) and natural phenomena (such as disease, predation, or changes in the distribution and abundance of their prey in response to changing climate) rather than endogenous threats caused by the small size of their population.

Diving and Social Behavior

In Hawai'ian waters, humpback whales remain almost exclusively within the 1820 m isobath and usually within waters depths less than 182 meters. Maximum diving depths are approximately 150 m (492 ft) (but usually <60 m [197 ft]), with a very deep dive (240 m [787 ft]) recorded off Bermuda (Hamilton et al. 1997). They may remain submerged for up to 21 min (Dolphin 1987). Dives on feeding grounds ranged from 2.1-5.1 min in the north Atlantic (Goodyear unpublished manuscript). In southeast Alaska average dive times were 2.8 min for feeding whales, 3.0 min for non-feeding whales, and 4.3 min for resting whales (Dolphin 1987). In the Gulf of California

humpback whale dive times averaged 3.5 min (Strong 1989). Because most humpback prey is likely found above 300 m depths most humpback dives are probably relatively shallow.

In a review of the social behavior of humpback whales, Clapham (1986) reported that they form small, unstable social groups during the breeding season. During the feeding season they form small groups that occasionally aggregate on concentrations of food. Feeding groups are sometimes stable for long-periods of times. There is good evidence of some territoriality on feeding (Clapham 1994, 1996), and calving areas (Tyack 1981). In calving areas, males sing long complex songs directed towards females, other males or both. The breeding season can best be described as a floating lek or male dominance polygyny (Clapham 1996). Intermale competition for proximity to females can be intense as expected by the sex ratio on the breeding grounds which may be as high as 2.4:1.

Vocalizations and Hearing

Humpback whales produce at least three kinds of vocalization: (1) complex songs with components ranging from at least 20 Hz to 4 kHz with estimated source levels from 144 to 174 dB, which are mostly produced by males on breeding areas (Payne 1970, Winn et al. 1970, Richardson et al. 1995); (2) social sounds in breeding areas that extend from 50 Hz to more than 10 kHz with most energy below 3 kHz (Tyack and Whitehead 1983, Richardson et al. 1995); and (3) vocalizations in foraging areas that are less frequent, but tend to be 20 Hz to 2 kHz with estimated sources levels in excess of 175 dB re 1 μPa-m (Thompson et al. 1986, Richardson et al. 1995). Sounds that investigators associate with aggressive behavior in male humpback whales are very different from songs; they extend from 50 Hz to 10 kHz (or higher), with most energy in components below 3 kHz (Tyack 1983, Silber 1986). These sounds appear to have an effective range of up to 9 kilometers (Tyack and Whitehead 1983).

In summary, humpback whales produce at least three kinds of sounds:

1. Complex songs with components ranging from at least 20 Hz–4 kHz with estimated source levels from 144 – 174 dB; these are mostly sung by males on the breeding grounds (Frazer and Mercado 2000; U.S. Navy 2006a; Payne 1970; Winn et al. 1970a; Richardson et al. 1995)
2. Social sounds in the breeding areas that extend from 50 Hz to more than 10 kHz with most energy below 3 kHz (Tyack and Whitehead 1983, Richardson et al. 1995); and
3 Feeding area vocalizations that are less frequent, but tend to be 20 Hz–2 kHz with estimated sources levels in excess of 175 dB re 1 μPa-m (Thompson et al. 1986; Richardson et al. 1995).

Helwig et al. (2000) produced a mathematical model of a humpback whale's hearing sensitivity based on the anatomy of the whale's ear. Based on that model, they concluded that humpback whales would be sensitive to sound in frequencies ranging from 0.7 kHz to 10k Hz, with a maximum sensitivity between 2 and 6k Hz. A general description of the anatomy of the ear for cetaceans is provided in the description of the fin whale above; that description is also applicable to humpback whales.

Sperm Whale

Distribution

Sperm whales occur in every ocean except the Arctic Ocean. Sperm whales are found throughout the North Pacific and are distributed broadly from tropical and temperate waters to the Bering Sea as far north as Cape Navarin. Mature, female, and immature sperm whales of both sexes are found in more temperate and tropical waters from the equator to around 45˚ N throughout the year. These groups of adult females and immature sperm whales are rarely found at latitudes higher than 50˚ N and 50˚ S (Reeves and Whitehead 1997). Sexually mature males join these groups throughout the winter. During the summer, mature male sperm whales are thought to move north into the Aleutian Islands, Gulf of Alaska, and the Bering Sea.

In the western Atlantic Ocean, sperm whales are distributed in a distinct seasonal cycle, concentrated east-northeast of Cape Hatteras in winter and shifting northward in spring when whales are found throughout the Mid-Atlantic Bight. Distribution extends further northward to areas north of Georges Bank and the Northeast Channel region in summer and then south of New England in fall, back to the Mid-Atlantic Bight.

In the eastern Atlantic Ocean, mature male sperm whales have been recorded as far north as Spitsbergen (Oien, 1990). Recent observations of sperm whales and stranding events involving sperm whales from the eastern North Atlantic suggest that solitary and paired mature male sperm whales predominantly occur in waters off Iceland, the Faroe Islands, and the Norwegian Sea (Gunnlaugsson and Sigurjonsson 1990, Oien 1990, Christensen et al. 1992).

In the Mediterranean Sea sperm whales are found from the Alboran Sea to the Levant Basin, mostly over steep slope and deep offshore waters. Sperm whales are rarely sighted in the Sicilian Channel, and are vagrant in the northern Adriatic and Aegean Seas (Notarbartolo di Sciara and Demma 1997). In the Italian seas sperm whales are more frequently associated with the continental slope off western Liguria, western Sardinia, northern and eastern Sicily, and both coasts of Calabria.

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Sperm whales commonly concentrate around oceanic islands in areas of upwelling, and along the outer continental shelf and mid-ocean waters. Because they inhabit deeper pelagic waters, their distribution does not include the broad continental shelf of the Eastern Bering Sea and these whales generally remain offshore in the eastern Aleutian Islands, Gulf of Alaska, and the Bering Sea.

Sperm whales have a strong preference for the 3,280 feet (1,000 meters) depth contour and seaward. Berzin (1971) reported that they are restricted to waters deeper than 300 meters (984 feet), while Watkins (1977) and Reeves and Whitehead (1997) reported that they are usually not found in waters less than 1,000 meters (3,281 feet) deep. While deep water is their typical habitat, sperm whales have been observed near Long Island, New York, in water between 41-55 meters (135-180 feet; Scott and Sadove 1997). When they are found relatively close to shore, sperm whales are usually associated with sharp increases in bottom depth where upwelling occurs and biological production is high, implying the presence of a good food supply (Clarke 1956).

Population Structure

The population structure of sperm whales is largely unknown. Lyrholm and Gyllenstein (1998) reported moderate, but statistically significant, differences in sperm whale mitochondrial (mtDNA) between ocean basins, although sperm whales throughout the world appear to be homogenous genetically (Whitehead 2003). Genetic studies also suggest that sperm whales of both genders commonly move across over ocean basins and that males, but not females, often breed in ocean basins that are different from the one in which they were born (Whitehead, 2003).

Sperm whales may not form “populations” as that term is normally conceived. Jaquet (1996) outlined a hierarchical social and spatial structure that includes temporary clusters of animals, family units of 10 or 12 females and their young, groups of about 20 animals that remain together for hours or days, “aggregations” and “super-aggregations” of 40 or more whales, and “concentrations” that include 1,000 or more animals (Peterson 1986, Whitehead and Wiegart 1990, Whitehead et al. 1991). The “family unit” forms the foundation for sperm whale society and most females probably spend their entire life in the same family unit (Whitehead 2002). The dynamic nature of these relationships and the large spatial areas they are believed to occupy might complicate or preclude attempts to apply traditional population concepts, which tend to rely on group fidelity to geographic distributions that are relatively static over time.

Atlantic Ocean

Based on harvests of tagged sperm whales or sperm whales with other distinctive marking, sperm whales in the North Atlantic Ocean appear to represent a single population, with the possible exception of the sperm whales that appear to reside in the Gulf of Mexico. Mitchell (1975) reported one sperm whale that was tagged on the Scotian Shelf and killed about 7 years later off Spain. Donovan (1991) reported five to six handheld harpoons from the Azore sperm whale fishery that were recovered from whales killed off northwest Spain, with another Azorean harpoon recovered from a male sperm whale killed off Iceland (Martin 1982). These patterns suggest that at least some sperm whales migrate across the North Atlantic Ocean.

Female and immature animals stay in Atlantic temperate or tropical waters year round. In the western North Atlantic, groups of female and immature sperm whales concentrate in the Caribbean Sea (Gosho et al. 1984) and south of New England in continental-slope and deep-ocean waters along the eastern United States (Blaylock et al. 1995). In eastern Atlantic waters, groups of female and immature sperm whales aggregate in waters off the Azores, Madeira, Canary, and Cape Verde Islands (Tomilin 1967).

Several investigators have suggested that the sperm whales that occupy the northern Gulf of Mexico are distinct from sperm whales elsewhere in the North Atlantic Ocean (Schmidly 1981, Fritts 1983, and Hansen et al. 1995), although the IWC groups do not treat these sperm whales as a separate population or "stock."

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Bayed and Beaubrun (1987) suggested that the frequent observation of neonates in the Mediterranean Sea and the scarcity of sperm whale sightings from the Gibraltar area may be evidence of a resident population of sperm whales in the Mediterranean.

Indian Ocean

In the Northern Indian Ocean the IWC recognized differences between sperm whales in the northern and southern Indian Ocean (Donovan 1991). Little is known about the Northern Indian Ocean population of sperm whales (Perry et al. 1999).

Several authors have proposed population structures that recognize at least three sperm whales populations in the North Pacific for management purposes (Kasuya 1991, Bannister and Mitchell 1980). At the same time, the IWC's Scientific Committee designated two sperm whale stocks in the North Pacific: a western and eastern stock or population (Donovan 1991). The line separating these populations has been debated since their acceptance by the IWC's Scientific Committee. For stock assessment purposes, NMFS recognizes three discrete population centers of sperm whales in the Pacific: (1) Alaska, (2) California-Oregon-Washington, and (3) Hawai'i.

Sperm whales are widely distributed throughout the Hawai'ian Islands throughout the year and are the most abundanct large whale in waters off Hawai'i during the summer and fall (Rice 1960, Shallenberger 1981, Lee 1993, and Mobley et al. 2000). Sperm whale clicks recorded from hydrophones off Oahu confirm the presence of sperm whales near the Hawai'ian Islands throughout the year (Thompson and Friedl 1982). The primary area of occurrence for the sperm whale is seaward of the shelf break in the Hawai'ian Islands.

Sperm whales have been sighted in the Kauai Channel, the Alenuihaha Channel between Maui and the island of Hawai'i, and off the island of Hawai'i (Lee 1993, Mobley et al. 1999, Forney et al. 2000). Additionally, the sounds of sperm whales have been recorded throughout the year off Oahu (Thompson and Friedl 1982). Twenty-one sperm whales were sighted during aerial surveys conducted in Hawai'ian waters conducted from 1993 through 1998. Sperm whales sighted during the survey tended to be on the outer edge of a 50 - 70 km distance from the Hawai'ian Islands, indicating that presence may increase with distance from shore. However, from the results of

these surveys, NMFS has calculated a minimum abundance of sperm whales within 46 km of Hawai'i to be 43 individuals (Forney et al. 2000).

Sperm whales south of the equator are generally treated as a single "population," although the IWC divides these whales into nine different divisions that are based more on evaluations of whaling captures than the biology of sperm whales (Donovan 1991). Several authors, however, have argued that the sperm whales that occur off the Galapagos Islands, mainland Ecuador, and northern Peru are geographically distinct from other sperm whales in the Southern Hemisphere (Rice 1977, Wade and Gerrodette 1993, and Dufault and Whitehead 1995).

Threats to the Species

Natural threats. Sperm whales are hunted by killer whales (Orcinus orca), false killer whales (Pseudorca crassidens), and short-finned pilot whales (Globicephala melas; Arnbom et al. 1987, Palacios and Mate 1996, Rice 1989, Weller et al. 1996, Whitehead 1995). Sperm whales have been observed with bleeding wounds on their heads and tail flukes after attacks by these species (Arnbom et al. 1987, Dufault and Whitehead 1995). In October 1997, 25 killer whales were documented to have attacked a group of mature sperm whales off Point Conception, California (personal communication from K Roberts cited in Perry et al. 1999) and successfully killing one of these mature sperm whales. Sperm whales have also been reported to have papilloma virus (Lambertson et al. 1987).

Studies on sperm whales in the North Pacific and North Atlantic Oceans have demonstrated that sperm whales are infected by calciviruses and papillomavirus (Smith and Latham 1978, Lambertsen et al. 1987). In some instances, these diseases have been demonstrated to affect 10 percent of the sperm whales sampled (Lambertsen et al. 1987).

Anthropogenic threats. Three human activities are known to threaten sperm whales: whaling, entanglement in fishing gear, and shipping. Historically, whaling represented the greatest threat to every population of sperm whales and was ultimately responsible for listing sperm whales as an endangered species. Sperm whales were hunted all over the world during the 1800s, largely for its spermaceti oil and ambergris. Harvesting of sperm whales subsided by 1880 when petroleum replaced the need for sperm whale oil (Whitehead 2003).

The actual number of sperm whales killed by whalers remains unknown and some of the estimates of harvest numbers are contradictory. Between 1800 and 1900, the IWC estimated that nearly 250,000 sperm whales were killed globally by whalers. From 1910 to 1982, another 700,000 sperm whales were killed globally by whalers (IWC Statistics 1959-1983). These estimates are substantially higher than a more recent estimate produced by Caretta et al. (2005), however, who estimated that at least 436,000 sperm whales were killed by whalers between 1800 and 1987. Hill and DeMaster (1999) concluded that about 258,000 sperm whales were harvested in the North Pacific between 1947 and 1987 by commercial whalers. They reported that catches in the North Pacific increased until 1968, when 16,357 sperm whales were harvested, then declined after 1968 because of harvest limits imposed by the IWC. Perry et al. (1999) estimated that, on average, more than 20,000 sperm whales were harvested in the Southern Hemisphere each year between 1956 and 1976.

These reports probably underestimate the actual number of sperm whales that were killed by whalers, particularly because they could not have incorporated realistic estimates of the number of sperm whales killed by Soviet whaling fleets, which often went unreported. Between 1947 and 1973, Soviet whaling fleets engaged in illegal whaling in the Indian, North Pacific, and southern Oceans. In the Southern Hemisphere, these whalers killed an estimated 100,000 whales that they did not report to the IWC (Yablokov et al. 1998). Illegal catches in the Northern Hemisphere (primarily in the North Pacific) were smaller but still caused sperm whales to disappear from large areas of the North Pacific Ocean (Yablokov and Zemsky 2000).

In addition to large and illegal harvests of sperm whales, Soviet whalers had disproportionate effect on sperm whale populations because they commonly killed adult females in any reproductive condition (pregnant or lactating) as well as immature sperm whales of either gender.

When the IWC introduced the International Observer Scheme in 1972, the IWC relaxed regulations that limited the minimum length of sperm whales that could be caught from 11.6 meters to 9.2 meters out of a concern that too many male sperm whales were being caught so reducing this size limit would encourage fleets to catch more females. Unfortunately, the IWC's decision had been based on data from the Soviet fleets who commonly reported female sperm whales as males. As a result, the new regulations allowed the Soviet whalers to continue their harvests of female and immature sperm whales legally, with substantial consequences for sperm whale populations. Berzin noted in a report he wrote in 1977, "the result of this was that some breeding areas for sperm whales became deserts" (Berzin 2007).

Although the IWC protected sperm whales from commercial harvest in 1981, whaling operations along the Japanese coast continued to hunt sperm whales in the North Pacific until 1988 (Reeves and Whitehead 1997). More recently, the Japanese Whaling Association began hunting sperm whales for research. In 2000, the Japanese Whaling Association announced that it planned to kill 10 sperm whales in the Pacific Ocean for research, which was the first time sperm whales have been hunted since the international ban on commercial whaling. Despite protests from the U.S. government and members of the IWC, the Japanese government harvested 5 sperm whales and 43 Bryde's whales in the last six months of 2000. According to the Japanese Institute of Cetacean Research (Institute of Cetacean Research undated), another 5 sperm whales were killed for research in 2002 – 2003. The consequences of these deaths on the status and trend of sperm whales remains uncertain, given that they probably have not recovered from the legacy of whaling; however, the renewal of a program that intentionally targets and kills sperm whales before we can be certain they recovered from a history of over-harvest places this species at risk in the foreseeable future.

Sperm whales are still hunted for subsistence purposes by whalers from Lamalera, Indonesia, which is on the south coast of the island of Lembata and from Lamakera on the islands of Solor. These whalers hunt in a traditional manner: with bamboo spears and using small wooden outriggers, 10–12 m long and 2 m wide, constructed without nails and with sails woven from palm fronds. The animals are killed by the harpooner leaping onto the back of the animal from the boat to drive in the harpoon. The maximum number of sperm whales killed by these hunters in any given year was 56 sperm whales killed in 1969.

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 9 sperm whales per year from 1991 - 1995 (Barlow et al. 1997). Interactions between longline fisheries and sperm whales in the Gulf of Alaska have been reported over the past decade (Rice 1989, Hill and DeMaster 1999). 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 1998). 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.

Sperm whales are also killed by ship strikes. In May 1994 a sperm whale that had been struck by a ship was observed south of Nova Scotia (Reeves and Whitehead 1997) and in May 2000 a merchant ship reported a strike in Block Canyon (NMFS, unpublished data), which is a major pathway for sperm whales entering southern New England continental shelf waters in pursuit of migrating squid (CeTAP 1982, Scott and Sadove 1997).

Status

Sperm whales were listed as endangered under the ESA in 1973. Sperm whales have been protected from commercial harvest by the IWC since 1981, although the Japanese continued to harvest sperm whales in the North Pacific until 1988 (Reeves and Whitehead 1997). They are also protected by the Convention on International Trade in Endangered Species of Wild Flora and Fauna and the MMPA. Critical habitat has not been designated for sperm whales.

The status and trend of sperm whales at the time of this summary is largely unknown. Hill and DeMaster (1999) and Angliss and Lodge (2004) reported that estimates for population abundance, status, and trends for sperm whales off the coast of Alaska were not available when they prepared the Stock Assessment Report for marine mammals off Alaska. Similarly, No information was available to support estimates of sperm whales status and trends in the western North Atlantic Ocean (Waring et al. 2004), the Indian Ocean (Perry et al. 1999), or the Mediterranean Sea.

Nevertheless, several authors and organizations have published "best estimates" of the global abundance of sperm whales or their abundance in different geographic areas. Based on historic whaling data,190,000 sperm whales were estimated to have been in the entire North Atlantic, but the IWC considers data that produced this estimate unreliable (Perry et al. 1999). Whitehead (2002) estimated that prior to whaling sperm whales numbered around 1,110,000 and that the current global abundance of sperm whales is around 360,000 (coefficient of variation = 0.36) whales. Whitehead's current population estimate (2002) is about 20% of past global abundance estimates which were based on historic whaling data.

Waring et al. (2007) concluded that the best estimate of the number of sperm whales along the Atlantic coast of the U.S. was 4,029 (coefficient of variation = 0.38) in 1998 and 4,804

(coefficient of variation = 0.38) in 2004, with a minimum estimate of 3,539 sperm whales in the western North Atlantic Ocean.

Barlow and Taylor (2005) derived two estimates of sperm whale abundance in a 7.8 million km^2 study area in the northeastern temperate Pacific: when they used acoustic detection methods they produced an estimate of 32,100 sperm whales (coefficient of variation = 0.36); when they used visual surveys, they produced an estimate of 26,300 sperm whales (coefficient of variation = 0.81). Caretta et al. (2005) concluded that the most precise estimate of sperm whale abundance off California, Oregon, and Washington was 1,233 (coefficient of variation = 0.41; based on ship surveys conducted in the summer and fall of 1996 and 2001). Their best estimate of the abundance of sperm whales in Hawai'i was 7,082 sperm whales (coefficient of variation = 0.30) based on ship-board surveys conducted in 2002.

Mark and recapture data from sperm whales led Whitehead and his co-workers to conclude that sperm whale numbers off the Galapagos Islands decreased by about 20% a year between 1985 and 1995 (Whitehead et al. 1997). In 1985 Whitehead et al. (1997) estimated there were about 4,000 female and immature sperm whales, whereas in 1995 they estimated that there were only a few hundred. They suggested that sperm whales migrated to waters off the Central and South American mainland to feed in productive waters of the Humboldt Current, which had been depopulated of sperm whales as a result of intensive whaling.

The information available on the status and trend of sperm whales do not allow us to make definitive statement about the extinction risks facing sperm whales as a species or particular populations of sperm whales. However, the evidence available suggests that sperm whale populations probably exhibit the dynamics of small populations, causing their population dynamics to become a threat in and of itself. The number of sperm whales killed by Soviet whaling fleets in the 1960s and 1970s would have substantial and adverse consequence for sperm whale populations and their ability to recover from the effects of whaling on their population. The number of adult female killed by Soviet whaling fleets, including pregnant and lactating females whose death would also have resulted in the death of their calves, would have had a devastating effect on sperm whale populations. In addition to decimating their population size, whaling would have skewed sex ratios in their populations, created gaps in the age structure of their populations, and would have had lasting and adverse effect on the ability of these populations to recover (for example, see Whitehead 2003).

Populations of sperm whales could not have recovered from the overharvests of adult females and immature whales in the 30 to 40 years that have passed since the end of whaling, but the information available does not allow us to determine whether and to what degree those populations might have stabilized or whether they have begun the process of recovering from the effects of whaling. Absent information to the contrary, we assume that sperm whales will have elevated extinction probabilities because of both exogenous threats caused by anthropogenic activities (primarily whaling, entanglement, and ship strikes) and natural phenomena (such as disease, predation, or changes in the distribution and abundance of their prey in response to changing climate) as well as endogenous threats caused by the legacy of overharvests of adult females and immature whales on their populations (that is, a population with a disproportion of

adult males and older animals coupled with a small percentage of juvenile whales that recruit into the adult population).

Diving and Social Behavior

Sperm whales are probably the deepest and longest diving mammal: they can dive to depths of at least 2000 meters (6562 ft), and may remain submerged for an hour or more (Watkins et al. 1993). Typical foraging dives last 40 min and descend to about 400 m followed by about 8 min of resting at the surface (Gordon 1987; Papastavrou et al. 1989). However, dives of over 2 hr and as deep as 3,000 m have been recorded (Clarke 1976; Watkins et al. 1985). Descent rates recorded from echo-sounders were approximately 1.7m/seconds and nearly vertical (Goold and Jones 1995). There are no data on diurnal differences in dive depths in sperm whales. However, like most diving vertebrates for which there are data (e.g. rorqual whales, fur seals, chinstrap penguins), sperm whales probably make relatively shallow dives at night when organisms from the ocean's deep scattering layers move toward the ocean's surface.

The groups of closely related females and their offspring develop dialects specific to the group (Weilgart and Whitehead 1997) and females other than birth mothers will guard young at the surface (Whitehead 1996) and will nurse young calves (Reeves and Whitehead 1997).

Vocalizations and Hearing

Sperm whales produce loud broad-band clicks from about 0.1 to 20 kHz (Weilgart and Whitehead 1993, 1997; Goold and Jones 1995). These have source levels estimated at 171 dB re 1 μPa (Levenson 1974). Current evidence suggests that the disproportionately large head of the sperm whale is an adaptation to produce these vocalizations (Norris and Harvey 1972; Cranford 1992; but see Clarke 1979). This suggests that the production of these loud low frequency clicks is extremely important to the survival of individual sperm whales. The function of these vocalizations is relatively well-studied (Weilgart and Whitehead 1993, 1997; Goold and Jones 1995). Long series of monotonous regularly spaced clicks are associated with feeding and are thought to be produced for echolocation. Distinctive, short, patterned series of clicks, called codas, are associated with social behavior and intragroup interactions; they are thought to facilitate intra-specific communication, perhaps to maintain social cohesion with the group (Weilgart and Whitehead 1993).

A general description of the anatomy of the ear for cetaceans is provided in the description of the fin whale above. The only data on the hearing range of sperm whales are evoked potentials from a stranded neonate (Carder and Ridgway 1990). These data suggest that neonatal sperm whales respond to sounds from 2.5-60 kHz. Sperm whales have been observed to frequently stop echolocating in the presence of underwater pulses made by echosounders and submarine sonar (Watkins and Schevill 1975; Watkins et al. 1985). 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). Sperm whales have moved out of areas after the start of air gun seismic testing (Davis et al. 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 1999). Furthermore, because of their apparent role as important predators of mesopelagic squid and fish, changing the abundance of sperm whales should affect the distribution and abundance of other marine species.

Steller Sea Lion

Distribution

Steller sea lions' range includes portions of the Gulf of Alaska. Steller sea lions are distributed mainly around the coasts to the outer continental shelf along the North Pacific Ocean rim from northern Hokkaiddo, Japan through the Kuril Islands and Okhotsk Sea, Aleutian Islands and central Bering Sea, southern coast of Alaska and south to California. The population is divided into the Western and the Eastern distinct population segments (DPSs) at 144° West longitude (Cape Suckling, Alaska). The Western DPS includes Steller sea lions that reside in the central and western Gulf of Alaska, Aleutian Islands, as well as those that inhabit the coastal waters and breed in Asia (e.g., Japan and Russia). The Eastern DPS includes sea lions living in southeast Alaska, British Columbia, California, and Oregon.

Steller sea lions do not migrate, but they often disperse widely outside of the breeding season (Loughlin 2002). Steller sea lions are gregarious animals that often travel or haul out in large groups of up to 45 individuals (Keple 2002). At sea, groups usually consist of females and subadult males; adult males are usually solitary while at sea (Loughlin 2002). An area of high occurrence extends from the shore to the 273-fathom (500-m) depth. For the Gulf of Alaska, foraging habitat is primarily shallow, nearshore, and continental shelf waters 4.3 to 13 nm (8 to 24 km) offshore with a secondary occurrence inshore of the 3,280 ft (1,000 m) isobath, and a rare occurrence seaward of the 3,280 ft (1,000 m) isobath. Steller sea lions have been sighted foraging in the middle of the Gulf of Alaska (Navy 2006). The April 2009 survey in the Gulf of Alaska TMAA encountered two groups of Steller sea lions (Rone et al. 2010).

Western DPS Population Structure

The minimum abundance estimate for the Western DPS of Steller sea lions is 38,988 individuals (Angliss and Allen 2009). Given the wide dispersal of individuals, the Western DPS may occur in the Gulf of Alaska (Angliss and Outlaw 2008; Navy 2006; NMFS 2008b), with about 70 percent of the population living in Alaskan waters. Between 2000 and 2004, the Western DPS increased at a rate of approximately 3 percent per year (Fritz and Stinchcomb 2005). Despite incomplete surveys conducted in 2006 and 2007, the available data indicate that the Western Steller sea lion DPS has been stable since 2004 (when the last complete assessment was done). The revised Steller Sea Lion Recovery Plan (NMFS 2008b) contains recovery criteria to change the listing of the Western DPS from endangered to threatened ("down-listing") and to remove it from the list of species requiring ESA protection (delist).

Eastern DPS Population Structure

The minimum abundance estimate for the Eastern DPS is estimated at 45,095 to 55,832 (Angliss and Allen 2009). The Eastern DPS has increased at an annual rate of approximately 3 percent since at least the late 1970s (Pitcher et al. 2007) and may be a candidate for removal from the list of threatened and endangered species (NMFS 2008b).

Natural Threats. Reproductive failure and neonate, juvenile, and adult mortality resulting from disease probably occur in both DPSs of Steller sea lions. Antibodies to two types of bacteria (Leptospira and Chlamydiia), one marine calicivirus (San lilipel Sea Lion Virus), and seal herpes

virus (SeHV), which could produce such effects, were present in blood taken from Steller sea lions in Alaska (Barlough et al. 1987; Calkins and Goodwin 1988; Vedder et al. 1987).

Causes of pup mortality include drowning, starvation caused by separation from the mother, crushing by larger animals, disease, predation, and biting by females other than the mother (Edie 1977; Orr and Poulter 1967). Pup mortality on rookeries has not been thoroughly studied.

Steller sea lions are probably eaten by killer whales and sharks, but the possible impact of these predators is unknown. The occurrence of shark predation on other North Pacific pinnipeds has been documented, but not well quantified (Ainley et al. 1985).

Parasites of Steller sea lions include intestinal cestodes; trematodes in the intestine and bile duct of the liver; nematodes in the stomach, intestine, and lungs; acainthocephalans in the intestine; acarian mites in the nasopharynx and lungs; and an anopluran skin louse(Dailey and Brownell 1972; Dailey and Hill 1970).

Anthropogenic Threats. Historically, the Eastern DPS of Steller sea lions was subjected to substantial mortality by humans, primarily due to commercial exploitation and both sanctioned and unsanctioned predator control (NMFS 2008b). Commercial exploitation occurred primarily in the 1800s and early 1900s while unsanctioned predator control probably persisted into the 1970s in some locations. State sanctioned commercial harvest of Steller sea lions ended in 1972 with the advent of the MMPA.

Although not well documented, there is little doubt that numbers of Steller sea lions were greatly reduced in many locations by these activities (NMFS 2008b). Commercial hunting and predator control activities have been discontinued and no longer affect this DPS. In contrast to the Western DPS, which is experiencing potential human-related threats from competition with fisheries (potentially high), incidental take by fisheries (low), and toxic substances (medium) no threats to continued recovery were identified for the Eastern DPS. Although several factors affecting the Western DPS also affect the Eastern DPS (e.g., environmental variability, killer whale predation, toxic substances, disturbance, shooting), these threats do not appear to be at a level sufficient to keep the Eastern DPS from continuing to recover, given the long term sustained growth of the population as a whole (NMFS 2008b).

Western DPS Status and Trend

The Steller sea lion was initially listed as a threatened species under the ESA on April 5, 1990 (55 FR 12645). In 1997, based on demographic and genetic dissimilarities, we designated two DPSs of Steller sea lions under the ESA: A Western DPS and an Eastern DPS (62 FR 24345, 62 FR 30772). Due to persistent decline, the Western DPS was reclassified as endangered. The Western DPS includes animals at and west of Cape Suckling, Alaska (144°W).
The Steller sea lion is designated as depleted under MMPA.

Eastern DPS Status and Trend
The Steller sea lion was initially listed as a threatened species under the ESA on April 5, 1990 (55 FR 12645). The Eastern DPS includes animals east of Cape Suckling (Angliss and Outlaw 2005; Loughlin 2002; NMFS 2008b) that extend into southeastern Alaska, and Canada. Rookeries of the Eastern DPS occur along the coasts of Oregon and California (NMFS 2008b). The Steller sea lion is designated as depleted under MMPA. A final revised species recovery plan addresses both the Western and Eastern DPSs (NMFS 2008b).
On December 13, 2010 NMFS published a 90-day finding on petitions to delist the Eastern DPS of the Steller sea lion. The finding stated that substantial scientific or commercial information is available such that a status review is warranted.

Diving and Social Behavior
Steller sea lions tend to make shallow dives of less than 820 ft (250 m) but are capable of deeper dives (NMFS 2008b). Adult females stay with their pups for a few days after birth before beginning a regular routine of alternating foraging trips at sea with nursing their pups on land. Female Steller sea lions use smell and distinct vocalizations to recognize and create strong social bonds with their newborn pups. Females usually mate again with males within 2 weeks after giving birth.

Vocalization and Hearing
On land, territorial male Steller sea lions usually produce low frequency roars (Loughlin et al. 1987; Schusterman et al. 1970). The calls of females range from 30 Hz to 3 kHz, with peak frequencies from 150 Hz to 1 kHz; typical duration is 1.0 to 1.5 sec (Campbell et al. 2002). Pups produce bleating sounds.

The only study involving underwater hearing of male and female steller sea lions was conducted by Kastelein et al (2005). The authors estimated male steller sea lion maximum hearing sensitivity at 1 kHz (77 dB received level) and best hearing (defined at 10 dB from the maximum sensitivity) between 1 and 16 kHz with poorer sensitivity below 1 kHz and above 16 kHz. The female steller sea lion's maximum sensitivity occurred at 25 kHz (at 73 dB received level) with poorer sensitivity below 16 kHz and above 25 kHz. However, because of the small number of animals tested, the findings could not be attributed to individual differences in sensitivity or sexual dimorphism (Kastelein et al. 2005). Because of the scarcity of information relating to hearing in steller sea lions and other pinnipeds, Southall et al (2007) estimated the functional underwater hearing range of all pinnipeds as between 75 Hz and 75 kHz.

Leatherback Sea Turtle
Distribution
Leatherback turtles are widely distributed throughout the oceans of the world. The species is found in four main regions of the world: the Pacific, Atlantic, and Indian Oceans, and the Caribbean Sea. Leatherbacks also occur in the Mediterranean Sea, although they are not known to nest there. The four main regional areas may further be divided into nesting aggregations. Leatherback turtles are found on the western and eastern coasts of the Pacific Ocean, with nesting aggregations in Mexico and Costa Rica (eastern Pacific) and Malaysia, Indonesia, Australia, the Solomon Islands, Papua New Guinea, Thailand, and Fiji (western Pacific). In the Atlantic Ocean, leatherback nesting aggregations have been documented in Gabon, Sao Tome

and Principe, French Guiana, Suriname, and Florida. In the Caribbean, leatherbacks nest in the U.S. Virgin Islands and Puerto Rico. In the Indian Ocean, leatherback nesting aggregations are reported in India and Sri Lanka.

Leatherback sea turtles are highly migratory, exploiting convergence zones and upwelling areas in the open ocean, along continental margins, and in archipelagic waters (Morreale et al. 1994, Eckert 1998, Eckert 1999a). In a single year, a leatherback may swim more than 10,000 kilometers (Eckert 1998). In the North Atlantic Ocean, leatherback turtles regularly occur in deep waters (>328 ft), and an aerial survey study in the north Atlantic sighted leatherback turtles in water depths ranging from 3 to 13,618 ft, with a median sighting depth of 131.6 ft (CeTAP 1982). This same study found leatherbacks in waters ranging from 7 to 27.2°C. In the Pacific Ocean, leatherback turtles have the most extensive range of any living reptile and have been reported in all pelagic waters of the Pacific between 71°N and 47°S latitude and in all other major pelagic ocean habitats (NMFS and USFWS 1998). Leatherback turtles lead a completely pelagic existence, foraging widely in temperate waters except during the nesting season, when gravid females return to tropical beaches to lay eggs. Males are rarely observed near nesting areas, and it has been hypothesized that leatherback sea turtles probably mate outside of tropical waters, before females swim to their nesting beaches (Eckert and Eckert 1988).

Leatherback turtles are uncommon in the insular Pacific Ocean, but individual leatherback turtles are sometimes encountered in deep water and prominent archipelagoes. To a large extent, the oceanic distribution of leatherback turtles may reflect the distribution and abundance of their macroplanktonic prey, which includes medusae, siphonophores, and salpae in temperate and boreal latitudes (NMFS and USFWS 1996). There is little information available on their diet in subarctic waters.

Population Structure

Leatherback turtles are widely distributed throughout the oceans of the world. The species is divided into four main populations in 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 populations are further 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 the Andaman and Nicobar Islands.

Threats to the Species

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

Anthropogenic threats. Leatherback sea turtles are endangered by several human activities, including fisheries interactions, entanglement in fishing gear (e.g., gillnets, longlines, lobster pots, weirs), direct harvest, egg collection, the destruction and degradation of nesting and coastal habitat, boat collisions, and ingestion of marine debris (NMFS and USFWS 1997).

Impacts of human activity on this species

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

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

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

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

Leatherback sea turtles are known to drown in fish nets set in coastal waters of Sao Tome, West Africa (Castroviejo et al. 1994; Graff 1995). Gillnets are one of the suspected causes for the decline in the leatherback turtle population in French Guiana (Chevalier et al. 1999), and gillnets targeting green and hawksbill turtles in the waters of coastal Nicaragua also incidentally catch leatherback turtles (Lagueux et al. 1998). Observers on shrimp trawlers operating in the northeastern region of Venezuela documented the capture of six leatherbacks from 13,600 trawls (Marcano and Alio, 2000). An estimated 1,000 mature female leatherback turtles are caught annually off of Trinidad and Tobago with mortality estimated to be between 50-95% (Eckert and Lien, 1999). However, many of the turtles do not die as a result of drowning, but rather because the fishermen butcher them in order to get them out of their nets (NMFS 2001). There are known to be many sizeable populations of leatherbacks nesting in West Africa, possibly as many as 20,000 females nesting annually (Fretey 2001). In Ghana, nearly two thirds of the leatherback turtles that come up to nest on the beach are killed by local fishermen.

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

Status

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

Globally, leatherback turtle populations have been decimated worldwide. In 1980, the global leatherback population was estimated at approximately 115,000 adult females (Pritchard 1982). By 1995, this global population (of adult females) is estimated to have declined to 34,500 (Spotila et al. 1996). Populations have declined in Mexico, Costa Rica, Malaysia, India, Sri Lanka, Thailand, Trinidad, Tobago, and Papua New Guinea. Throughout the Pacific, leatherbacks are seriously declining at all major nesting beaches.

In the Atlantic and Caribbean, the largest nesting assemblages of leatherbacks are found in the U.S. Virgin Islands, Puerto Rico, and Florida. Since the early 1980s, nesting data has been collected at these locations. Populations in the eastern Atlantic (i.e. off Africa) and Caribbean appear to be stable; however, information regarding the status of the entire leatherback population in the Atlantic is lacking and it is certain that some nesting populations (e.g., St. John and St. Thomas, U.S. Virgin Islands) have been extirpated (NMFS and USFWS 1995). Data

collected in southeast Florida clearly indicate increasing numbers of nests for the past twenty years (9.1-11.5% increase), although it is critical to note that there was also an increase in the survey area in Florida over time (NMFS 2001). However, the largest leatherback rookery in the western North Atlantic remains along the northern coast of South America in French Guiana and Suriname. Recent information suggests that Western Atlantic populations declined from 18,800 nesting females in 1996 (Spotila et al. 1996) to 15,000 nesting females by 2000 (Spotila, personal communication cited in NMFS 2001). The nesting population of leatherback turtles in the Suriname-French Guiana trans-boundary region has been declining since 1992 (Chevalier and Girondot, 1998). Poaching and fishing gear interactions are believed to be the major contributors to the decline of leatherbacks in the area.

Leatherback sea turtles appear to be in a critical state of decline in the North Pacific Ocean. The leatherback population that nests along the east Pacific Ocean was estimated to be over 91,000 adults in 1980 (Spotila 1996), but is now estimated to number less than 3,000 total adult and subadult animals (Spotila 2000). Leatherback turtles have experienced major declines at all major Pacific basin rookeries. At Mexiquillo, Michoacan, Mexico, Sarti et al. (1996) reported an average annual decline in nesting of about 23% between 1984 and 1996. The total number of females nesting on the Pacific coast of Mexico during the 1995-1996 season was estimated at fewer than 1,000. Less than 700 females are estimated for Central America (Spotila 2000). In the western Pacific, the decline is equally severe. Current nestings at Terengganu, Malaysia represent 1% of the levels recorded in the 1950s (Chan and Liew 1996).

While Spotila et al. (1996) indicated that turtles may have been shifting their nesting from French Guiana to Suriname due to beach erosion, analyses show that the overall area trend in number of nests has been negative since 1987 at a rate of 15.0 -17.3 % per year (NMFS 2001). If turtles are not nesting elsewhere, it appears that the Western Atlantic portion of the population is being subjected to mortality beyond sustainable levels, resulting in a continued decline in numbers of nesting females.

Based on published estimates of nesting female abundance, leatherback populations are declining at all major Pacific basin nesting beaches, particularly in the last two decades (Spotila et al. 1996, NMFS and USFWS 1998, Spotila et al. 2000). Declines in nesting populations have been documented through systematic beach counts or surveys in Malaysia (Rantau Abang, Terengganu), Mexico and Costa Rica. In other leatherback nesting areas, such as Papua New Guinea, Indonesia, and the Solomon Islands, there have been no systematic consistent nesting surveys, so it is difficult to assess the status and trends of leatherback turtles at these beaches. In all areas where leatherback nesting has been documented, however, current nesting populations are reported by scientists, government officials, and local observers to be well below abundance levels of several decades ago. The collapse of these nesting populations was most likely precipitated by a tremendous overharvest of eggs coupled with incidental mortality from fishing (Sarti et al. 1996, Eckert, 1997).

Based on recent modeling efforts, some authors concluded that leatherback turtle populations cannot withstand more than a 1% human-related mortality level which translates to 150 nesting females (Spotila et al. 1996). As noted previously, there are many human-related sources of mortality to leatherbacks; every year, 1,800 leatherback turtles are expected to be captured or

killed as a result of federally-managed activities in the U.S. (this total includes both lethal and non-lethal take). An unknown number of leatherbacks are captured or killed in fisheries managed by states. Spotila et al. (1996) recommended not only reducing fishery-related mortalities, but also advocated protecting eggs and hatchlings. Zug and Parham (1996) point out that a combination of the loss of long-lived adults in fishery-related mortalities and a lack of recruitment stemming from elimination of annual influxes of hatchlings because of intense egg harvesting has caused the sharp decline in leatherback populations.

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

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

Diving and Social Behavior

The maximum dive depths for post-nesting female leatherbacks in the Caribbean have been recorded at 475 meters and over 1,000 meters, with routine dives recorded at between 50 and 84 meters. The maximum dive length recorded for such female leatherback turtles was 37.4 minutes, while routine dives ranged from 4 -14.5 minutes (in Lutcavage and Lutz 1997). Leatherback turtles also appear to spend almost the entire portion of each dive traveling to and from maximum depth, suggesting that maximum exploitation of the water column is of paramount importance to the leatherback (Eckert et al. 1989).

A total of six adult female leatherback turtles from Playa Grande, Costa Rica were monitored at sea during their internesting intervals and during the 1995 through 1998 nesting seasons. The turtles dived continuously for the majority of their time at sea, spending 57 - 68% of their time submerged. Mean dive depth was 19 1 meters and the mean dive duration was 7.40.6 minutes (Southwood et al. 1999). Similarly, Eckert (1999) placed transmitters on nine leatherback females nesting at Mexiquillo Beach and recorded dive behavior during the nesting season. The majority of the dives were less than 150 meters depth, although maximum depths ranged from 132 meters to over 750 meters. Although the dive durations varied between individuals, the majority of them made a large proportion of very short dives (less than two minutes), although

Eckert (1999) speculates that these short duration dives most likely represent just surfacing activity after each dive. Excluding these short dives, five of the turtles had dive durations greater than 24 minutes, while three others had dive durations between 12 - 16 minutes.

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

Hearing

There is no information on leatherback sea turtle hearing. However, we assume that their hearing sensitivities will be similar to those of green and loggerhead sea turtles: their best hearing sensitivity will be in the low frequency range: from 200 to 400 Hz with rapid declines for tones at lower and higher frequencies. Their hearing will probably have a practical upper limit of about 1000 Hz (Bartol et al. 1999, Ridgway et al. 1969).

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

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 biological opinion includes the effects of several activities that affect the survival and recovery of endangered whales in the action area.

A number of human activities have contributed to the current status of populations of large whales, steller sea lions and sea turtles in the action area. Some of those activities, most notably commercial whaling, occurred extensively in the past, ended, and no longer appear to affect these whale populations, although the effects of these reductions likely persist today. Other human activities are ongoing and appear to continue to affect populations of endangered and threatened whales, sea turtles and salmon. The following discussion summarizes the principal phenomena that are known to affect the likelihood that these endangered and threatened species will survive and recover in the wild.

Natural Mortality
Natural mortality rates in cetaceans and pinnipeds, especially large whale species, are largely unknown. Although factors contributing to natural mortality cannot be quantified at this time, there are a number of suspected causes, including parasites, predation and red tide toxins. For example, the giant spirurid nematode (*Crassicauda boopis*) has been attributed to congestive kidney failure and death in some large whale species (Lambertsen 1986). A well-documented observation of killer whales attacking a blue whale off Baja, California proves that blue whales are at least occasionally vulnerable to these predators (Tarpy 1979). Other stochastic events, such as fluctuations in weather and ocean temperature affecting prey availability, may also contribute to large whale natural mortality.

For Steller sea lions, neonate, juvenile, and adult mortality result from disease caused by bacteria (Leptospira and Chlamydiia, etc. and viruses (seal herpes virus). Causes of pup mortality include drowning, starvation caused by separation from the mother, crushing by larger animals, disease, predation, and biting by females other than the mother (Edie 1977; Orr and Poulter 1967). Pup mortality on rookeries has not been thoroughly studied. Predation by killer whales and sharks also occur but the possible impact of these predators is unknown. Parasites of Steller sea lions include intestinal cestodes; trematodes in the intestine and bile duct of the liver; nematodes in the stomach, intestine, and lungs; acainthocephalans in the intestine; acarian mites in the nasopharynx and lungs; and an anopluran skin louse(Dailey and Brownell 1972; Dailey and Hill 1970).

Sea turtles are also affected by disease and environmental factors. Turtles can be injured by predators such as birds, fish, and sharks (George 1997). Hypothermic or cold stunning occurs when a turtle is exposed to cold water for a period of time. Cold stunned turtles often have decreased salt gland function which may lead to plasma electrolyte imbalance and a lowered immune response (George 1997).

Human-Induced Mortality
Large whale population numbers in the proposed action areas have historically been impacted by commercial exploitation, mainly in the form of whaling. Prior to current prohibitions on whaling, such as the International Whaling Commission's 1966 moratorium, most large whale species had been depleted to the extent it was necessary to list them as endangered under the Endangered Species Act of 1966. For example, from 1900 to 1965 nearly 30,000 humpback whales were captured and killed in the Pacific Ocean with an unknown number of additional animals captured and killed before 1900 (Perry et al. 1999a). Sei whales are estimated to have been reduced to 20 percent (8,600 out of 42,000) of their pre-whaling abundance in the North Pacific (Tillman 1977). In addition, 9,500 blue whales were reported killed by commercial whalers in the North Pacific between 1910-1965 (Ohsumi and Wada. 1972); 46,000 fin whales between 1947-1987 (Rice 1984); and 25,800 sperm whales (Barlow et al. 1997). North Pacific right whales once numbered 11,000 animals but commercial whaling has now reduced their population to 29-100 animals (Wada 1973).

Entrapment and entanglement in commercial fishing gear is one of the most frequently documented sources of human-caused mortality in large whale species, pinnipeds and sea turtles. For example, in 1978, Nishimura and Nakahigashi (1990) estimated that 21,200 turtles, including

greens, leatherback turtles, loggerheads, olive ridleys and hawksbills, were captured annually by Japanese tuna longliners in the Western Pacific and South China Sea, with a reported mortality of approximately 12,300 turtles per year. Using commercial tuna longline logbooks, research vessel data and questionnaires, Nishimura and Nakahigashi (1990) estimated that for every 10,000 hooks in the Western Pacific and South China Sea, one turtle is captured, with a mortality rate of 42 percent.

Western Steller sea lions are also captured in fishing gear. For example, the Gulf of Alaska Pollock trawl has captured between 24 to 31 steller sea lions each year from 2002 to 2006 with 2 observed mortalities (Allen and Angliss 2011). Preliminary data from 2007 and 2008 indicate that no stellers were captured in this fishery. The mean annual rate of mortality from 2002-2008 is 1.33 animals per year (CV – 0.66).

NMFS has observed 3,251 sets, representing approximately 3,874,635 hooks (data from February 1994 through December 31, 1999). The observed entanglement rate for sperm whales would equal about 0.31 whales per 1,000 sets or 0.0002 per 1,000 hooks. At those rates, we would expect about 200 sperm whales entanglements per 1,000 sets. However, only one sperm whale has been entangled in this gear; as a result, NMFS believes that the estimated entanglement rate substantially overestimates a sperm whale's actual probability of becoming entangled in this gear and the potential hazards longline gear poses to sperm whales.

Collisions with commercial ships are an increasing threat to many large whale species, particularly as shipping lanes cross important large whale breeding and feeding habitats or migratory routes. The number of observed physical injuries to humpback whales as a result of ship collisions has increased in Hawaiian waters (Glockner-Ferrari et al. 1987). On the Pacific coast, a humpback whale is probably killed about every other year by ship strikes (Barlow *et al*. 1997). From 1996-2002, eight humpback whales were reported struck by vessels in Alaskan waters. In 1996, a humpback whale calf was found stranded on Oahu with evidence of vessel collision (propeller cuts; NMFS unpublished data). From 1994 to 1998, two fin whales were presumed to have been killed in ship strikes.

Despite these reports, the magnitude of the risks ship traffic poses to large whales on or around the Gulf of Alaska is difficult to quantify or estimate. We struggle to estimate the number of whales that are killed or seriously injured in ship strikes within the territorial seas and the Exclusive Economic Zone of the continental United States and have virtually no information on interactions between ships and commercial vessels in the western North Pacific Ocean. With the information available, we assume that interactions occur but we cannot estimate the number of interactions or their significance to the endangered whales of the western North Pacific Ocean.

Chronic exposure to the neurotoxins associated with paralytic shellfish poisoning (PSP) via zooplankton prey has been shown to have detrimental effects on marine mammals. Estimated ingestion rates are sufficiently high to suggest that the PSP toxins are affecting marine mammals, possibly resulting in lower respiratory function, changes in feeding behavior and lower reproduction fitness (Durbin et al. 2002). Other human activities, including discharges from wastewater systems, dredging, ocean dumping and disposal, aquaculture and additional impacts from coastal development are also known to impact marine mammals and their habitat. In the

North Pacific, undersea exploitation and development of mineral deposits, as well as dredging of major shipping channels pose a continued threat to the coastal habitat of right whales. Point-source pollutants from coastal runoff, offshore mineral and gravel mining, at-sea disposal of dredged materials and sewage effluent, potential oil spills, as well as substantial commercial vessel traffic, and the impact of trawling and other fishing gear on the ocean floor are continued threats to marine mammals in the proposed action area.

The impacts from these activities are difficult to measure. However, some researchers have correlated contaminant exposure to possible adverse health effects in marine mammals. Studies of captive harbor seals have demonstrated a link between exposure to organochlorines (*e.g.*, DDT, PCBs, and polyaromatic hydrocarbons) and immunosuppression (De Swart et al. 1996; Harder et al. 1992; Ross et al. 1995). Organochlorines are chemicals that tend to bioaccumulate through the food chain, thereby increasing the potential of indirect exposure to a marine mammal via its food source. During pregnancy and nursing, some of these contaminants can be passed from the mother to developing offspring. Contaminants like organochlorines do not tend to accumulate in significant amounts in invertebrates, but do accumulate in fish and fish-eating animals. Thus, contaminant levels in planktivorous mysticetes have been reported to be one to two orders of magnitude lower compared to piscivorous odontocetes (O'Hara and Rice 1996; O'Hara et al. 1999; O'Shea and Brownell Jr. 1994).

The effects of climate change on marine species in the action area remain largely unknown. Gaps in information on species movements and distribution, the difficulty involved with studying highly mobile animals such as marine mammals and turtles, as well as insufficient historical information and long-term data sets on habitat and distribution all complicate any potential conclusions on the effects of climate change for these species (Kintisch 2006; Simmonds and Isaac 2007). However, possible effects of climatic variability include the following: alteration of ecological community composition and structure, possibly resulting in species relocating from areas they currently use in response to changes in oceanic conditions; use of an altered range as temperature-dependent distribution limits change; changes to migration patterns or community structure; changes to species abundance; increased susceptibility to disease and contaminants; alterations to prey composition and availability; and altered timing of breeding (MacLeod *et al.* 2005; Robinson *et al.* 2005; Kintisch 2006; Learmonth *et al.* 2006; McMahon and Hays 2006). Such changes could affect reproductive success and survival, and therefore have consequences for the recovery of both large whales, pinnipeds and sea turtles (Robinson *et al.* 2005; Learmonth *et al.* 2006; Cotté and Guinet 2007).

The marine mammals and sea turtles that occur in the action area are regularly exposed to several sources of natural and anthropogenic sounds. Anthropogenic noises that could affect ambient noise arise from the following general types of activities in and near the sea, any combination of which can contribute to the total noise at any one place and time. These noises include transportation, dredging, construction; oil, gas, and mineral exploration in offshore areas; geophysical (seismic) surveys; sonars; explosions; and ocean research activities (Richardson et al. 1995).

Noise in the marine environment has received a lot of attention in recent years and is likely to continue to receive attention in the foreseeable future. Several investigators have argued that

anthropogenic sources of noise have increased ambient noise levels in the ocean over the last 50 years (Jasny et al. 2005; NRC 1994; NRC 2000; NRC 2003; NRC 2005; Richardson et al. 1995). Much of this increase is due to increased shipping as ships become more numerous and of larger tonnage (NRC 2003). Commercial fishing vessels, cruise ships, transport boats, airplanes, helicopters and recreational boats all contribute sound into the ocean (NRC 2003). The military uses sound to test the construction of new vessels as well as for naval operations. In some areas where oil and gas production takes place, noise originates from the drilling and production platforms, tankers, vessel and aircraft support, seismic surveys, and the explosive removal of platforms (NRC 2003). Many researchers have described behavioral responses of marine mammals to the sounds produced by helicopters and fixed-wing aircraft, boats and ships, as well as dredging, construction, geological explorations, etc. (Richardson et al. 1995). Most observations have been limited to short-term behavioral responses, which included cessation of feeding, resting, or social interactions. Several studies have demonstrated short-term effects of disturbance on humpback whale behavior (Baker et al. 1983; Bauer and Herman 1986; Hall 1982; Krieger and Wing 1984), but the long-term effects, if any, are unclear or not detectable. Carretta et al.(2001)and Jasny et al. (2005) identified the increasing levels of anthropogenic noise as a habitat concern for whales and other cetaceans because of its potential effect on their ability to communicate.

Surface shipping is the most widespread source of anthropogenic, low frequency (0 to 1,000 Hz) noise in the oceans (Simmonds and Hutchinson. 1996). The radiated noise spectrum of merchant ships ranges from 20 to 500 Hz and peaks at approximately 60 Hz. Ross (1976) has estimated that between 1950 and 1975 shipping had caused a rise in ambient ocean noise levels of 10 dB. He predicted that this would increase by another 5 dB by the beginning of the 21st century.

Ambient Noise

Urick (1983) provided a discussion of the ambient noise spectrum expected in the deep ocean. Shipping, seismic activity, and weather are primary causes of deep-water ambient noise. Noise levels between 20 and 500 Hz appear to be dominated by distant shipping noise that usually exceeds wind-related noise. Above 300 Hz, the level of wind-related noise might exceed shipping noise. Wind, wave, and precipitation noise originating close to the point of measurement dominate frequencies from 500 to 50,000 Hz. The ambient noise frequency spectrum and level can be predicted fairly accurately for most deep-water areas based primarily on known shipping traffic density and wind state (wind speed, Beaufort wind force, or sea state) (Urick 1983). For frequencies between 100 and 500 Hz, Urick (1983) has estimated the average deep water ambient noise spectra to be 73 to 80 dB for areas of heavy shipping traffic and high sea states, and 46 to 58 dB for light shipping and calm seas.

In contrast to deep water, ambient noise levels in shallow waters (i.e., coastal areas, bays, harbors, etc.) are subject to wide variations in level and frequency depending on time and location. The primary sources of noise include distant shipping and industrial activities, wind and waves, and marine animals (Urick 1983). At any given time and place, the ambient noise level is a mixture of these noise types. In addition, sound propagation is also affected by the variable shallow water conditions, including the depth, bottom slope, and type of bottom. Where the bottom is reflective, the sound levels tend to be higher than when the bottom is absorptive.

In addition to the federal vessel operations, private and commercial shipping vessels, vessels (both commercial and private) engaged in marine mammal watching also have the potential to impact whales in the proposed action area. A recent study of whale watch activities worldwide has found that the business of viewing whales and dolphins in their natural habitat has grown rapidly over the past decade into a billion dollar ($US) industry involving over 80 countries and territories and over 9 million participants (Hoyt 2001).

Several investigators have studied the effects of whale watch vessels on marine mammals (Amaral and Carlson 2005; Au and Green 2000; Corkeron 1995; Erbe 2002b; Felix 2001; Magalhaes et al. 2002; Richter et al. 2006; Scheidat et al. 2004; Simmonds 2005; Watkins 1986; Williams et al. 2002). The whale's behavioral responses to whale watching vessels depended on the distance of the vessel from the whale, vessel speed, vessel direction, vessel noise, and the number of vessels. The whales' responses changed with these different variables and, in some circumstances, the whales did not respond to the vessels, 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.

The Impact of the Baseline on Listed Resources

Although listed resources are exposed to a wide variety of past and present state, Federal or private actions and other human activities that have already occurred or continue to occur in the action area as well as Federal projects in the action area that have already undergone formal or early section 7 consultation, and State or private actions that are contemporaneous with this consultation, the impact of those activities on the status, trend, or the demographic processes of threatened and endangered species remains largely unknown.

Historically, commercial whaling had occurred in the action area and had caused all of the large whales to decline to the point where the whales faced risks of extinction that were high enough to list them as endangered species. Since the end of commercial whaling, the primary threat to these species has been eliminated. However, all of the whale species have not recovered from those historic declines and scientists cannot determine if those initial declines continue to influence current populations of most large whale species. Species like North Pacific right whales have not begun to recover from the effects of commercial whaling on their populations and continue to face very high risks of extinction in the foreseeable future because of their small population sizes (on the order of 50 individuals) and low population growth rates. Relationships between potential stressors in the marine environments and the responses of these species that may keep their populations depressed are unknown.

Recent attention has focused on the emergence of a wide number of anthropogenic sound sources and their role as a pollutant in the marine environment. Relationships between specific sound sources, or anthropogenic sound generally, and the responses of marine mammals to those sources are still subject to extensive scientific research and public inquiry but no clear patterns have emerged.

Few of the anthropogenic phenomena in the Gulf of Alaska that represent potential risks to whales in the Action Area seem likely to kill whales. Instead, most of these phenomena — close approaches by whale-watching and research vessels, anthropogenic sound sources, pollution, and

many fishery interactions — would affect the behavioral, physiological, or social ecology of whales in the region. Reports suggest that the response of whales to many of the anthropogenic activities in the Action Area are probably short-lived, which suggests that the responses would not be expected to affect the fitness of individual whales. Most of these reports relate to humpback whales during their winter, breeding season; there are very few reports of the behavioral responses of other whale species to human activity in the action area.

Gauthier and Sears (1999), Weinrich et al. (1992), Clapham and Mattila (1993), Clapham et al. (1993) concluded that close approaches for biopsy samples or tagging did not cause humpback whales to respond or caused them to exhibit "minimal" responses when approaches were "slow and careful." This caveat is important and is based on studies conducted by Clapham and Mattila (1993) of the reactions of humpback whales to biopsy sampling in breeding areas in the Caribbean Sea. These investigators concluded that the way a vessel approaches a group of whales had a major influence on the whale's response to the approach; particularly cow and calf pairs. Based on their experiments with different approach strategies, they concluded that experienced, trained personnel approaching humpback whales slowly would result in fewer whales exhibiting even a minimal response.

At the same time, several lines of evidence suggest that these human activities might have greater consequences for individual whales (if not for whale populations). Several investigators reported behavioral responses to close approaches that suggest that individual whales might experience stress responses. Baker et al. (1983) described two responses of whales to vessels, including: (1) "horizontal avoidance" of vessels 2,000 to 4,000 meters away characterized by faster swimming and fewer long dives; and (2) "vertical avoidance" of vessels from 0 to 2,000 meters away during which whales swam more slowly, but spent more time submerged. Watkins (1981c) found that both fin and humpback whales appeared to react to vessel approach by increasing swim speed, exhibiting a startled reaction, and moving away from the vessel with strong fluke motions.
Bauer (1986) and Bauer and Herman (1986) studied the potential consequences of vessel disturbance on humpback whales wintering off Hawai′i. They noted changes in respiration, diving, swimming speed, social exchanges, and other behavior correlated with the number, speed, direction, and proximity of vessels. Results were different depending on the social status of the whales being observed (single males when compared with cows and calves), but humpback whales generally tried to avoid vessels when the vessels were 0.5 to 1.0 kilometer from the whale. Smaller pods of whales and pods with calves seemed more responsive to approaching vessels.

Baker et al. (1983) and Baker and Herman (1986) summarized the response of humpback whales to vessels in their summering areas and reached conclusions similar to those reached by Bauer and Herman (1986): these stimuli are probably stressful to the humpback whales in the action area, but the consequences of this stress on the individual whales remains unknown. Studies of other baleen whales, specifically bowhead and gray whales document similar 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. 1985). For example, studies of bowhead whales revealed that these whales oriented themselves in relation to a vessel when the engine was on, and exhibited significant avoidance responses when the vessel's engine was turned on even at a

distance of approximately 3,000 ft (900 m). Weinrich et al. (1992) associated “moderate” and “strong” behavioral responses with alarm reactions and stress responses, respectively.

Jahoda et al. (2003) studied the response of 25 fin whales in feeding areas in the Ligurian Sea to close approaches by inflatable vessels and to biopsy samples. They concluded that close vessel approaches caused these whales to stop feeding and swim away from the approaching vessel. The whales also tended to reduce the time they spent at surface and increase their blow rates, suggesting an increase in metabolic rates that might indicate a stress response to the approach. In their study, whales that had been disturbed while feeding remained disturbed for hours after the exposure ended. They recommended keeping vessels more than 200 meters from whales and having approaching vessels move at low speeds to reduce visible reactions in these whales.

Beale and Monaghan (2004a) concluded that the significance of disturbance was a function of the distance of humans to the animals, the number of humans making the close approach, and the frequency of the approaches. These results would suggest that the cumulative effects of the various human activities in the action area would be greater than the effects of the individual activity. None of the existing studies examined the potential effects of numerous close approaches on whales or gathered information on levels of stress-related hormones in blood samples that are more definitive indicators of stress (or its absence) in animals.

There is mounting evidence that wild animals respond to human disturbance in the same way that they respond to predators (Beale and Monaghan 2004a; 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 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).

The information available does not allow us to assess the actual or probable effects of natural and anthropogenic phenomena on threatened or endangered species in the action area. The age composition, gender ratios, population abundance, and changes in that abundance over time remain unknown for threatened and endangered species in the action area of this consultation. Without this information or some surrogate information, it would be difficult, if not impossible, to reliably assess the impact of the activities identified in this *Environmental Baseline* on threatened and endangered species in the action area.

Effects of the Proposed Actions

Pursuant to Section 7(a)(2) of the ESA, federal agencies are directed to ensure that their activities are not likely to jeopardize the continued existence of any listed species or result in the destruction or adverse modification of critical habitat. The proposed survey and issuance of the IHA by NMFS for “takes” of marine mammals during the seismic studies would expose listed species to seismic airgun pulses, as well as sound emitted from a multi-beam bathymetric

echosounder and sub-bottom profiler. In this section, we describe the potential physical, chemical, or biotic stressors associated with the proposed actions, the probability of individuals of listed species being exposed to these stressors based on the best scientific and commercial evidence available, and the probable responses of those individuals (given probable exposures) based on the available evidence. As described in the *Approach to the Assessment* section, for any responses that would be expected reduce an individual's fitness (i.e., growth, survival, annual reproductive success, and lifetime reproductive success), the assessment would consider the risk posed to the viability of the population(s) those individuals comprise and to the listed species those populations represent. The purpose of this assessment is to determine if it is reasonable to expect the proposed activities to have effects on listed species that could appreciably reduce their likelihood of surviving and recovering in the wild.

For this consultation, we are particularly concerned about behavioral disruptions that may result in animals that fail to feed or breed successfully or fail to complete their life history because these responses are likely to have population-level consequences. The proposed IHA would authorize non-lethal "takes" by harassment of listed species during survey activities. The ESA does not define harassment nor has NMFS defined the term pursuant to the ESA through regulation. However, the Marine Mammal Protection Act of 1972, as amended, defines harassment as any act of pursuit, torment, or annoyance which has the potential to injure a marine mammal or marine mammal population in the wild or has the potential to disturb a marine mammal or marine mammal population 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 latter portion of this definition (that is, "...causing disruption of behavioral patterns including...migration, breathing, nursing, breeding, feeding, or sheltering") is almost identical to the U.S. Fish and Wildlife Service's regulatory definition of "harass"[4] pursuant to the ESA. For this Opinion, we define harassment similarly: an intentional or unintentional human act or omission that creates the probability of injury to an individual animal by disrupting one or more behavioral patterns that are essential to the animal's life history or its contribution to the population the animal represents.

Evidence Available for the Assessment

Given the nature of the proposed seismic activities, the effects of anthropogenic sound on ESA-listed whales, Steller sea lions and leatherback sea turtles are assessed in this consultation. Information on these effects is limited, and methods to acquire acoustic information, such as audiograms of large whales, are not available. The available information on hearing capabilities and mechanisms employed for receiving and interpreting sounds remains very limited due to the cryptic nature of some species and their rarity, the large size of many species, and the difficulties associated with performing field studies on these animals. Underwater hearing abilities have been studied experimentally in a few species. Where experimental data do not exist, some inference of the sound frequencies that are important to large whales and sea turtles can be made from the characteristics of the sounds they produce or from the physiology of their hearing organs.

4 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 behavior patterns which include, but are not limited to, breeding, feeding, or sheltering (50 CFR 17.3)

Assumptions

In conducting the effects analysis for the proposed actions, several assumptions must be made due to gaps in available information. Definitive statements on the effects of sound from the proposed activities are complicated because detection of sounds by these animals depends on the acoustic properties of the source (spectral characteristics and intensity), transmission characteristics of the water, and sensitivity of hearing in each species. Furthermore, responses to sounds can be highly variable between individuals and may depend on an animal's activity at time of exposure, motivation for that activity, age, and any habituation or sensitization to sounds.

The effects analysis in this Opinion reviews information on the characteristics of sounds resulting from the proposed action, incorporates assumptions about large whale, pinniped and sea turtle hearing abilities based on available information (as presented in the *Status of Listed Resources* section), and examines published studies of animals' responses upon exposure to sounds. Although the airguns, multi-beam bathymetric echosounder, and sub-bottom profiler would be operating simultaneously, the dominant sound at distances from the *Langseth* would be the low-frequency airguns given that transmission loss for higher-frequency sounds is relatively greater. Based on the assumptions for what the different taxa can hear, we expect that sperm whales would be sensitive to mid- to high-frequency sounds, large baleen whales (i.e., fin, and humpback whales) and sea turtles would be sensitive to lower frequency sounds such as those produced by seismic airguns and Steller sea lions would be sensitive to low and mid-frequency sounds. Published studies of these and other species' responses or lack of response to anthropogenic sounds are available; we assume that responses noted in these studies mean that individuals of similar species, for which no studies are available, would respond similarly. To examine the potential for sounds to mask the detection of natural sounds at similar frequencies, or to induce temporary or permanent reductions in an individual's hearing threshold, the analysis examined results from controlled exposure studies. Given fundamental similarities in ear anatomy among marine mammals, sperm whales were expected to experience similar types of physiological changes when exposed to similar sounds. There is no information regarding leatherback sea turtle hearing sensitivities. However, we assume that their hearing sensitivities will be similar to those of green and loggerhead sea turtles (their best hearing sensitivity will be in the low frequency range from 200 to 400 Hz with rapid declines for tones at lower and higher frequencies), and therefore, we will assume information related to other sea turtle species apply to leatherback sea turtles as well.

Potential Stressors

The assessment for this consultation identified several possible stressors associated with the proposed seismic activities: oil or fuel leakage from vessels or towed equipment, potential ship strikes, potential for entanglement or interaction with tail buoys, sounds generated by vessel engines, effects from the towed hydrophone streamer, effects from the OBSs, effects from the towed hydrophone streamer, and acoustic energy from airguns and sonar introduced into the marine environment during the seismic operations. Based on a review of available information, the consultation determined which of these possible stressors would be likely to occur and which would be negligible. Accordingly, this consultation focused on the following stressors that are likely to be produced by the proposed seismic activities and may affect ESA-listed species: (1) acoustic energy introduced into the marine environment by the airgun array, and (2) acoustic

energy introduced by both the multi-beam bathymetric echosounder (MBES) and sub-bottom profiler (SBP) sonar and the ocean bottom seismometers (OBS).

Airgun Array

Sounds produced by the airguns are short pulses occurring for less than one second. For the proposed activity, the pulse duration is 0.1s and would occur at 22 s for seismic surveying with the hydrophone streamer, or every 66 s when recording data on the OBSs as the *Langseth* travels 7.4–9.3 km/h. Most of the energy in the sound pulses emitted by airguns occurs at low frequencies (0-188 Hz), with considerably lower levels for frequencies above 1,000 Hz, and smaller amounts of energy emitted up to ~150 kHz (LGL Ltd. 2011).

As described in LGL, Ltd (2011), airguns function by venting high-pressure, compressed air into the water. The pressure signature of an individual airgun consists of a sharp rise and then fall in pressure, followed by several positive and negative pressure oscillations. The sizes, arrangement, and firing times of the individual airguns in an array are designed and synchronized to suppress the pressure oscillations subsequent to the first cycle, and coalesce pressure levels into one pulse. The resulting downward-directed pulse has a fraction of a second duration, with only one strong positive and one strong negative peak pressure (Caldwell and Dragoset 2000).

The strengths of airgun pulses can be measured in different ways. Peak-to-peak levels (pk-pk) are presented in units of dB re 1 μPa. The peak level (0-pk) for the same pulse is typically about 6 dB less. In the biological literature, levels of received airgun pulses are often described based on the "average" or "root-mean-square (rms)" level over the duration of the pulse. The rms value for a given pulse is typically about 10 dB lower than 0-pk, and 16 dB lower than pk-pk (McCauley *et al.* 2000b; Greene *et al.* 1997, McCauley *et al.* 1998 both *as cited in* NMFS 2006h). A fourth measure – sound energy level – is sometimes used and is expressed in dB re 1 $\mu Pa^2 \cdot s$; however, because seismic pulses are less than one second in duration, the numerical value of sound energy level is lower than the rms pressure level.

As mentioned in the *Description of the Proposed Actions* section, the 36-airgun array will have an estimated peak (0-pk) sound source level of 259 dB re 1 μPa and a peak to-peak of 265 dB. The source levels for airgun arrays are nominal source levels for sound directed downward, which represent the theoretical source level close to a single point source emitting the same sound as that emitted by the arrays. The actual source for airgun arrays is a distributed sound source (i.e., multiple guns) rather than a single point source. In order to communicate the levels of sound from the array, back calculations must be made from far field measurements to acquire a theoretical value of the source level. This theoretical source level is never actually realized for airgun arrays because airguns are distributed over several meters; the highest sound levels actually measurable in the water close to the airgun array (such as one meter) will never be as high as the nominal source level. For the *Langseth* array, the highest sound level actually measurable at any location in the water from the airguns is estimated at approximately 264 dB (pk-pk). Given that rms levels are typically 16 dB lower than pk-pk values, the estimated rms source level for the *Langseth* airgun array would be 248 dB (rms). In addition, because of the directional nature (downward) of the sound propagating from these airgun arrays, the effective source level for sound propagating in near-horizontal directions will be substantially lower than the nominal source level. The theoretical point source estimates for airgun arrays are useful,

however, for accurately reflecting received levels in the far-field (at more than 75 to 100 m; Caldwell and Dragoset 2000).

Pathways for received seismic sound include direct paths from the source, indirect paths that include reflection from the sea surface and bottom, and often indirect paths including segments propagating through bottom sediments. Sound propagating via indirect paths travels longer distances and often arrives later than sounds received via a direct path. However, sound may also travel faster through sediments than in water, and thus may arrive earlier than the direct arrival despite traveling a greater distance. Variations in travel time lengthen the duration of a received pulse; seismic pulses of about 10 to 20 ms in duration at the source can be longer when received at long horizontal distances. For example, for an airgun array operating in the Beaufort Sea, pulse duration was about 300 ms at a distance of 8 km, 500 ms at 20 km, and 850 ms at 73 km (Greene and Richardson 1988).

Multi-beam Bathymetric Echosounder (MBES)
The proposed activities would include multi-beam bathymetric echosounder (MBES) sonar, operated continuously, to map the ocean floor. The hull-mounted MBES has a narrow fore-aft beamwidth, operates at high frequencies (10.5–13 kHz), and has a maximum source level of 242 dB re 1 μPa.

For deep-water operation, each "ping" consists of eight (in water depths greater than 1000 m) or four (in water depths less than 1000 m) successive fan-shaped transmissions which ensonifies a sector that extends 1° fore-aft. The successive transmissions span an overall cross-track angular extent of about 150°, with 2-ms gaps between pings for successive sectors. Continuous wave signals (waves of constant energy and frequency) increase from 2 to 15 ms long in water depths up to 2600 m and frequency-modulated (FM) chirp signals up to 100 ms long are used in water depths greater than 2600 m.

Sub-Bottom Profiler (SBP)
The proposed survey would also use a Knudsen 320B sub-bottom profiler to provide information about the sea floor. The SBP would operate simultaneously with the airgun array and the MBES. Energy from the SBP is directed downward by a hull-mounted 3.5 kHz transducer and varies with water depth. The SBP has a maximum source level of 204 dB re 1 μPa but varies with water depth. Nominal beam width is 30 degrees, and pulse duration would be 1, 2, or 4 ms, with bandwidths of 1.0 kHz, 0.5 kHz, and 0.25 kHz, respectively. The interval between SBP pulses would be 1 s, with a common mode of operation being five pulses at 1-s intervals followed by a 5-s pause.

Ocean Bottom Seismometers (OBS)
The proposed action includes the deployment and retrieval of OBSs. Once the OBS is ready to be retrieved, an acoustic release transponder interrogates the OBS at a frequency of 9–11 kHz, and a response is received at a frequency of 9–13 kHz.

EXPOSURE ANALYSIS

Exposure analyses identify the ESA-listed species that are likely to co-occur with the actions' effects on the environment in space and time, and identify the nature of that co-occurrence. The

Exposure Analysis identifies, as possible, the number, age or life stage, and gender of the individuals likely to be exposed to the actions' effects and the population(s) or subpopulation(s) those individuals represent.

NMFS applies certain acoustic thresholds to help determine at what point during exposure to seismic airguns (and other acoustic sources) marine mammals are considered "harassed", pursuant to the MMPA (65 FR 16374; March 28, 2000). These thresholds are used to develop safety radii around a source and the necessary power-down or shut-down criteria, and are applied to sea turtles for the proposed activities as well. Seismic airgun noise can propagate substantial distances (e.g., Nieukirk *et al.* 2004), although at lower sound levels than the designated acoustic thresholds. L-DEO estimated the safety radii around the proposed *Langseth* operations using an acoustic propagation model, adjusted with empirical data gathered in the Gulf of Mexico in 2009. The modeled distances to which sound levels (rms) might be received in deep (> 1,000m) waters as in this survey for the 36-airgun array and the 40 in3 single airgun to be used during the proposed survey were provided in Table 1 on page 7.

The exposure analysis for this consultation is concerned with the numbers of fin, humpback and sperm whales and leatherback sea turtles likely to be exposed to received levels greater than 180 dB re 1 µPa (rms), which constitutes the shut-down criterion for cetaceans that is also applied here to sea turtles. The maximum distance from airguns where received levels might meet the shut-down criterion of 180 dB (i.e., from the full 36-gun array and the single airgun) are estimated to be as follows:

- 36 airgun array: 0.940 km (0.51 nm)
- 40 in^3 single airgun: 0.040 km (0.02 nm)

This exposure analysis for this consultation is concerned with the numbers of Steller sea lions likely to be exposed to received levels greater than 190 dB re 1 µPa (rms),which constitutes the shut-down criterion for pinnipeds. The maximum distance from airguns where received levels might meet the shut-down criterion of 190 dB (i.e., from the full 36-gun array and the single airgun) are estimated to be as follows:

- 36 airgun array: 0.40 km (0.22 nm)
- 40 in^3 single airgun: 0.012 km (0.01 nm)

These maximum distances all fall well within the visibility range from the *Langseth.* When stationed on the observation platform on the *Langseth*, eye level is about 21.5 m (70.5 ft) above sea level and PSVOs would be able to see around the entire vessel and to a distance of about 10 km with the naked eye, 5 km or further with the Big Eyes, 200 m with night vision and 2-3 km or further with the reticle binoculars. The 180 dB radius will not always reach these distances, as shorter radii will occur during the use of smaller numbers of airguns (e.g., the use of a single airgun during turns or power-down procedures). However, based on these maximum propagation distances, our concern is the probability of ESA-listed whales, Steller sea lions and leatherback sea turtles occurring within this range from the *R/V Langseth* during seismic operations.

Marine Mammals

The IHA application contained the estimated number of ESA-listed whales and Steller sea lions that might be exposed to received levels equal to or greater than 160 dB re 1 µPa (rms) and 170 dB re 1 µPa (rms), respectively in the action area (LGL Ltd. 2011). LGL, Ltd (2011) states that an individual mammal would not be exposed numerous times during the survey since the seismic lines are widely spaced in the survey area. The whale exposure estimates are based on the best available information on whale densities and a planned ensonified area of ~26,166 km^2 (the planned track lines plus the contingency track line) that would be within the 160-dB isopleth on one or more occasions during the survey. Steller sea lion exposure estimates are based on the best available information on Steller sea lion densities and a planned ensonified area of ~15,372.5 km^2 (the planned track lines plus the contingency track line) that would be within the 170-dB isopleth on one or more occasions during the survey.

The resulting best estimates includes no blue whales, 76 fin whales, 68 humpback whales, no sei whales, 10 sperm whales and 256 Steller sea lions (Table 2). LGL, Ltd, (2011) estimated the percent of the population that would be exposed to acoustic sounds greater than 160 dB dB re 1 µPa for whales and 170 dB re 1 µPa for Steller sea lions as listed in Table 2. These estimates are probably over-estimates as they assume that no animals would move away from the sounds. Blue whales, sei whales and North Pacific right whales are not expected to be exposed to survey activities because of their rarity in the Action area, however, USGS will shut down all acoustic sources if one of these species is sighted at any distance from the Langseth. Given this, if these species are exposed to survey activities we do not expect exposures of these species to acoustic sources at levels greater than 160 dB dB re 1 µPa.

Table 2. Estimated exposure of ESA-listed whales to sound levels $\geq$160 dB re 1 µPa and exposure of Steller sea lions to sound levels $\geq$170 dB re 1 µPa during the proposed survey activities in the central Gulf of Alaska.

Species	Number of Animals Exposed	Percent of Regional Population
Fin Whales	76	0.47%
Humpback Whales	68	0.33%
Sperm Whales	10	0.04%
Steller Sea Lions	256	0.61%

From the information available, we cannot estimate the age or life stage, gender, or reproductive condition of the individuals fin and humpback whales that might be exposed to survey activities. We assume these whales could represent any age class and either sex. Sperm whale would be subadult and adult males since females and juveniles of both genders generally do not migrate above around 40° north latitude. Steller sea lions would be male and female juveniles and adults from either the Eastern or Western DPS. As Steller sea lions can breed as early as two weeks after giving birth, we expect that breeding animals would also be present during survey activities.

Mitigation measures and monitoring activities during the proposed survey include visual and passive acoustic monitoring, an exclusion zone within the 180 dB isopleths for cetaceans and leatherback sea turtles, an exclusion zone within the 190 dB isopleths for Steller sea lions, speed

and course alterations when practicable, power-down and shut-down procedures, an emergency shut-down provision, and ramp-up procedures for airguns. These measures are expected to reduce the risk that ESA-listed whales and Steller sea lions would occur within the 180 dB radius and 190 dB radius, respectively; therefore, any exposures that might occur are more likely to involve fin, humpback or sperm whales at received levels less than 180 dB re 1 μPa and Steller sea lions at received levels less than 190 dB re 1 μPa.

Sea Turtles

A few experiments where sea turtles were exposed to airgun sounds (e.g., see Moein *et al.* 1994; McCauley *et al.* 2000a; 2000b) indicate that avoidance of seismic sources by sea turtles may likely occur. However, monitoring reports from seismic surveys in other regions (including the Hess Deep area of the ETP) suggest that some sea turtles did not avoid airguns and were likely exposed to higher levels of seismic airgun pulses (Smultea and Holst 2003 *as cited in* NMFS 2006h). Accordingly, we expect some sea turtles may be exposed to the proposed activities. We assume individual leatherback sea turtles in the Action Area would represent either sex and any marine stage. Female leatherback sea turtles would be members of either the Costa Rican or Mexican nesting population. We cannot estimate the proportions of individual female leatherback sea turtles in the Action Area belonging to each population. We cannot estimate the population affiliation of male leatherback sea turtles that might be exposed.

Mitigation measures and monitoring activities would also be applied for sea turtles during the proposed activities and include the same measures as applied for marine mammals. These measures are expected to reduce the risk that ESA-listed sea turtles would occur within the 180 dB radius; therefore, any exposures that might occur are more likely to involve leatherback sea turtles at received levels less than 180 dB re 1 μPa.

Exposure to MBES and SBP

The vessel-based sonar to be used during the proposed activities is downward-directed with a narrow fore-aft beamwidth. Any exposures to sonar pings that might occur would be expected to be brief, given the short duration of pulses and the fact the vessel will be transiting and ensonifying a narrow swath. The potential for exposure to these sources is further reduced by the measures in place to minimize exposure to seismic airguns within the 180 dB radius. Due to the directional nature of both the sub-bottom profiler and the echosounder, surface ducting of sound produced by this equipment is not expected to occur.

Sperm whales exhibit hearing at higher frequencies, and sound pulses from the multi-beam and sub-bottom sonar would be audible to individual sperm whales within the narrow extent of a transmitted sound beam. In addition, humpback whale hearing may extend into the mid-frequency range based on available information; therefore, the 3.5 kHz output of the sub-bottom profiler may be audible to this species as well. Based on the hearing abilities of Steller sea lions we would expect that sound pulses from the multi-beam and sub-bottom sonar would be audible to individuals of this species within the narrow extent of a transmitted sound beam. However, Kremser *et al.* (2005) concluded the probability of a cetacean swimming through the area of exposure when such sources emit a pulse is small, as the animal would have to pass at close range and be swimming at speeds similar to the vessel to be subjected to sound levels that could cause temporary threshold shifts. Similarly for Steller sea lions, we would expect the probability

of swimming through the area of exposure when such sources emit a pulse is also small. For sea turtles, and fin whales, available information indicates detection of sounds in the low-frequency range below those produced by the MBES or SBP equipment. Based on this information, these species exposed to received levels of mid-frequency (1 kHz–10 kHz) sounds or higher are not likely to detect these sounds, and therefore, will not respond to these sounds.

Exposure to OBS

Base on the hearing capabilities of cetaceans and pinnipeds and the frequencies of the sound pulses from the OBS, we expect that sperm whales as well as humpback whales and Steller sea lions would hear the acoustic release transponder and the response.

RESPONSE ANALYSIS

As discussed in the *Approach to the Assessment* section of this Opinion, response analyses determine how listed resources are likely to respond after being exposed to an action's effects on the environment or directly on listed species themselves. For the purposes of consultation, our assessments try to detect potential lethal, sub-lethal (or physiological), or behavioral responses that might result in reducing the fitness of listed individuals. Ideally, response analyses would consider and weigh evidence of adverse consequences as well as evidence suggesting the absence of such consequences.

Effects of exposure to airguns

A pulse of seismic airgun sound displaces water around the airgun and creates a wave of pressure, resulting in physical effects on the marine environment that can then affect marine organisms, such as the listed whales, Steller sea lions and sea turtles considered in this Opinion. Possible responses considered in this analysis consist of (1) threshold shifts; (2) auditory interference (masking); (3) behavioral responses; and (4) non-auditory physical or physiological effects. The *Response Analysis* also considers information on the potential for stranding and the potential effects on the prey of ESA-listed whales, Steller sea lions and leatherback sea turtles in the Action Area.

Threshold Shifts

Few studies exist that examine hearing impairment in marine mammals or sea turtles resulting from exposure to a strong sound. An animal can experience temporary threshold shift (TTS) or permanent threshold shift (PTS). A threshold shift involves reduced sensitivity to sounds, requiring them to be stronger to be audible. Duration of TTS can be minutes, hours, or days, with eventual recovery to normal hearing thresholds. For sound exposures near the TTS onset threshold, hearing sensitivity recovers rapidly after exposure to the noise ends (LGL, Ltd 2011); however, few data on sound levels and durations necessary for mild TTS have been obtained for marine mammals, and none of the published data examine TTS elicited by exposure to multiple pulses of sound (LGL, Ltd 2011). PTS involves physical damage to the sound receptors in the ear, resulting in total or partial deafness, or impairment of hearing at specific frequency ranges. There is no specific evidence that exposure to pulses of airgun sound can cause PTS in any marine mammal (Miller 2005).

Both TTS and PTS pose potential risks to marine mammals because they appear to extract a lot of information about their environment using hearing – e.g., information on the proximity of

predators, the distribution and abundance of prey, changes in weather patterns and oceanic conditions, and information on and from other members of their species, among other information. Reducing the ability of these whales to hear natural sounds could have potential adverse consequences for the fitness of individuals experiencing threshold shifts.

For sperm whales, LGL, Ltd (2008) concluded from the available data that when exposed to single short pulses, the TTS threshold appears to be a function of the energy content of the pulse (Finneran et al. 2002, 2005 as cited in LGL, Ltd 2008). Given the available data, the received energy level of a single seismic pulse (with no frequency weighting) might need to be 186 dB SEL or ~196–201 dB re 1 μPa (rms) in order to produce brief, mild TTS. Exposure to several strong seismic pulses that each have received levels near 190 dB re 1 μPa (rms) might result in cumulative exposure of ~186 dB SEL and, thus, slight TTS, assuming the TTS threshold is (to a first approximation) a function of the total received pulse energy. The distances from the Langseth's airguns at which the received energy level (per pulse, flat-weighted) would be expected to be greater than or equal to 190 dB re 1 μPa (rms) are estimated to be no more than 400 m (Table 1 page 7). For an odontocete closer to the surface, the maximum radius with greater than or equal to 190 dB re 1 μPa (rms) would be smaller. A higher level of sound is necessary to cause PTS. On an SEL basis, Southall et al. (2007) estimated that received levels would need to exceed the TTS threshold by at least 15 dB for there to be risk of PTS. Thus, for cetaceans they estimate that the PTS threshold might be an M-weighted SEL (for the sequence of received pulses) of ~198 dB re 1 μPa2 s (15 dB higher than the TTS threshold for an impulse), where the SEL value is cumulated over the sequence of pulses. A cetacean would need to received one or more pulses with peak pressure exceeding 230 or 218 dB re 1 μPa (0-peak), respectively. A peak pressure of 230 dB re 1 μPa (3.2 bar m, 0-pk) would only be found within a few meters of the largest (360-in3) airguns in the planned airgun array (Caldwell and Dragoset 2000).

For baleen whales, there are no studies to indicate the levels or properties of sound required to induce TTS. LGL, Ltd (2011) concluded that the frequencies to which baleen whales are most sensitive are assumed to be lower than those to which odontocetes are most sensitive, and natural background noise levels at those low frequencies tend to be higher. As a result, auditory thresholds of baleen whales within their frequency band of best hearing are believed to be higher (less sensitive) than are those of odontocetes at their best frequencies (Clark and Ellison 2004 as cited in LGL, Ltd 2011). From this, it is suspected that received levels causing TTS onset may also be higher in baleen whales (Southall et al. 2007).

We do not expect that sperm whales or baleen whales are likely to experience TTS or PTS from the proposed activities. Levels adjacent to the airguns may not be sufficient to induce PTS in whales, especially because an individual would not be exposed to more than one pulse at this received level unless it swam alongside the airgun for longer than the inter-pulse period. In addition, mitigation measures will be used, including avoidance of blue, sei and North Pacific right whales and concentrations of humpback and sperm whales, visual and acoustic monitoring, and ramp-up procedures for the array. These measures are expected to reduce the level of any exposures that may occur and further minimize the risk of PTS, as well as TTS.

In pinnipeds, TTS thresholds associated with exposure to brief pulses (single or multiple) of underwater sound have not been measured. Initial evidence from more prolonged (non-pulse)

exposures suggested that some pinnipeds (harbor seals in particular) incur TTS at somewhat lower received levels than do small odontocetes exposed for similar durations (Kastak et al. 1999, 2005; Ketten et al. 2001). The TTS threshold for pulsed sounds has been indirectly estimated as being an SEL of ~171 dB re 1 μPa^2 s (Southall et al. 2007), which would be equivalent to a single pulse with received level ~181–186 dB re 1 μPa(rms), or a series of pulses for which the highest rms values are a few dB lower. Mitigation measures mentioned above for whales would also apply to Steller sea lions and would help reduce the received level of any exposures that may occur and further minimize the risk of PTS or TTS.

Although sea turtles detect low frequency sound, the potential effects on sea turtle biology remain largely unknown (Samuel 2005). TTS in loggerhead sea turtles is reported to have been observed during studies by Moein *et al.* (1994). Turtle hearing was tested before, within 24 hours after, and two weeks after exposure to pulses of airgun sound. Moein *et al.* (1994) used an evoked potential method to test sea turtle hearing and concluded that the turtles exhibited some change in their hearing when tested within 24 hours after exposure (relative to pre-exposure hearing). The authors found that hearing had reverted to normal when tested two weeks after exposure. The size of the airgun used or the received sounds levels were not provided; therefore, the levels of airgun sounds that apparently elicited TTS are not known (NMFS 2006c). These findings indicate that TTS may occur in sea turtles exposed to seismic sources; however, turtles in the study were confined. Sea turtles at sea might exhibit avoidance behavior and, thus, reduced exposure to seismic pulses. For any sea turtles that exhibit little or no behavioural avoidance, or if turtles habituate to seismic noise such that avoidance reactions cease, these individuals could sustain hearing loss if exposed to high enough sound levels from seismic airguns (LGL Ltd. 2008). Mitigation measures mentioned above for whales and Steller sea lions would also apply to sea turtles and would help reduce the received level of any exposures that may occur and further minimize the risk of PTS or TTS.

Auditory Interference (Masking)

Interference, or masking, generally occurs when the interfering noise is of a similar frequency and louder than the auditory signal received by an animal processing echolocation signals or listening for acoustic information from other individuals. Generally, noise will only mask a signal if it is sufficiently close to the signal in frequency. Low frequency sounds are broad and tend to have relatively constant bandwidth, whereas higher frequency bandwidths are narrower (NMFS 2006h). It is probable that masking would be more likely to result from a continuous noise rather than short pulses (Richardson *et al.* 1995).

Limited data exist on the masking effects of pulsed sounds. Seismic sources emit short pulses lasting less than a second every 20 seconds or longer (e.g., 22 and 66 seconds for the proposed survey activities). The short duration and discontinuous nature of seismic pulses present a limited probability of masking natural sounds with low frequencies. Any masking that might occur would likely be temporary because seismic sources are discontinuous and the seismic vessel would continue to transit. The proposed seismic surveys could mask whale calls at some of the lower frequencies, in particular for baleen whales but also for sperm whales. This could affect communication between individuals, affect their ability to receive information from their environment, or affect sperm whale echolocation (Evans 1998; NMFS 2006h). Madsen *et al.* (2006) reported that, when oceanographic conditions were appropriate, sperm whales at the

surface appeared to receive seismic pulses containing higher frequencies (between 300 Hz to 3 kHz), although at much lower received levels. Most of the energy of sperm whales clicks is concentrated at 2 to 4 kHz and 10 to 16 kHz, and though the findings by Madsen *et al.* (2006) suggest frequencies of seismic pulses can overlap this range, the strongest spectrum levels of airguns are below 200 Hz (0-188 Hz for the *Langseth* airguns). Given the disparity between sperm whale echolocation and communication-related sounds with the dominant frequencies for seismic surveys, masking is not likely to be significant for sperm whales (NMFS 2006h). Overlap of the dominant low frequencies of airgun pulses with low-frequency baleen whale calls would be expected to pose a greater risk of effects due to masking. However, even for those frequencies that may overlap, the low duty cycle of airguns indicates that pulses are not likely to pose a significant masking problem for communication or echolocation in whales (Madsen *et al.* 2002; 2006).

Male Steller sea lions usually produce low frequency roars (Loughlin et al. 1987; Schusterman et al. 1970). The calls of females range from 30 Hz to 3 kHz, with peak frequencies from 150 Hz to 1 kHz; typical duration is 1.0 to 1.5 sec (Campbell et al. 2002). Although the findings by Madsen *et al.* (2006) suggest frequencies of seismic pulses can overlap the range of Steller sea lion communication, the strongest spectrum levels of airguns are below 200 Hz (0-188 Hz for the *Langseth* airguns). The proposed seismic survey could mask some Steller sea lion calls, however, any masking that might occur would likely be temporary because seismic sources are discontinuous and the seismic vessel would continue to transit.

For sea turtles, hearing capabilities are centered in the low frequency range, as are the dominant frequencies of seismic pulses. Given the overlap, it would not be unreasonable to anticipate some masking may result from the proposed seismic activities. Limited information is available on the potential for masking with respect to sea turtles; however, given that seismic pulses would last a fraction of a second at intervals of 22 or 66 seconds, any masking that may occur would be temporary as the seismic vessel transits through an area.

Behavioral Responses of Whales

Marine mammals may briefly respond to underwater sound by slightly changing their behavior or relocating a small distance, in which case the effects of these changes are unlikely to be significant to the individual, or by being displaced from important feeding or breeding areas over a prolonged period, in which case impacts on the individual could be significant. Marine mammal responses to anthropogenic sound vary by species, state of maturity, prior exposure, current activity, reproductive state, time of day, and other factors.

For fin, humpback and sperm whales that may be present in the action area and exposed to seismic airgun sounds at levels of 160 dB re 1μPa or higher, several field studies of the behavioral responses of these species, or lack of responses, have been conducted. Blue and fin whales have occasionally been reported in areas ensonified by airgun pulses; however, systematic data on their reactions to airgun sound are generally lacking. One study for blue whales off Oregon reported that whales continued vocalizing at the same rate as before exposure to airgun pulses, suggesting that at least their vocalization behavior was undisturbed by the sound (McDonald *et al.* 1993). Sightings by observers on seismic vessels off the U.K. suggest that during times of good sightability, the numbers of Balaenopterids (such as blue, fin, and

humpback whales) seen are similar between times when airguns are firing as well as silent (Stone 1997, 1998, 2000, 2001 *as cited in* NMFS 2006d). However, fin and sei whale sighting rates were higher when airguns were shooting, perhaps due to whales remaining at the surface at times of airgun operations (Stone 2003). The analysis of combined data from all years by Stone (2003 *as cited in* NMFS 2006d) indicated that baleen whales stayed farther from airguns, altered their course more often, and were headed away from the vessel more frequently during periods of shooting, suggesting some level of localized avoidance of seismic activity. Although information for blue and fin whales is limited, studies of other baleen whales are consistent (e.g., bowhead whales, see Miller 2005; Yazvenko *et al.* 2007) and indicate these species generally tend to avoid operating airguns, with avoidance radii being quite variable.

McCauley *et al.* (1998; 2000b) studied the responses of humpback whales off western Australia to a full-scale seismic survey with a 16-gun 2,678-in^3 array, as well as to a single 20-in^3 airgun with a source level of 227 db re 1μPa·m (pk-pk). The authors found the overall distribution of humpback whales migrating through the study area was unaffected by the full-scale seismic program, but that localized avoidance of the array and, to a lesser extent, the single airgun did occur. Avoidance reactions began at 5–8 km from the array and 2 km from the single airgun. Mean avoidance distance from the airgun corresponded to a received sound level of 140 db re 1μPa (rms), the level at which humpbacks started to show avoidance reactions to an approaching airgun. However, some humpback whales, especially males, approached within distances 100 to 400 m where the maximum received level was 179 db re 1μPa (rms). Potter *et al.* (2007) also reported localized avoidance of seismic airguns off Nova Scotia, but that whales did not move outside the detection range. Humpback whales summering in southeast Alaska did not exhibit persistent avoidance when exposed to seismic pulses from a 100-in^3 airgun (Malme *et al.* 1985 *as cited in* NMFS 2006d). Some humpbacks appeared to exhibit a startle response at received levels of 150-169 db re 1μPa. Despite the possibility of subtle effects, the author concluded there was no clear evidence of avoidance at received levels up to 172 db re 1μPa. These studies indicate that humpback whales could begin avoiding the proposed seismic survey at received lower received levels, but that some individuals may not avoid the airgun operations at levels up to 180 db re 1μPa.

Data on the short-term responses or lack of response by these whales to impulsive noise do not necessarily provide information about the long-term effects of such exposure. It is not known whether impulsive noises affect reproductive rate or distribution and habitat use in subsequent days or years. For other baleen whales, reports indicate that habitat use is not significantly altered in the long-term. Gray whales continue to migrate annually along the west coast of North America despite intermittent seismic exploration in that area for decades (Malme *et al.* 1984). Johnson *et al.* (2007) reported that gray whales exposed to seismic airguns off Sakhalin Island, Russia, did not experience any biologically significant or population level effects, based on subsequent research in the area from 2002–2005. Bowhead whales continued to travel to the eastern Beaufort Sea each summer despite seismic exploration in their summer and autumn range for many years. This species was often seen in summering areas where seismic exploration occurred during preceding summers (Richardson *et al.* 1986). Bowhead whales have also been observed over periods of days or weeks in areas repeatedly ensonified by seismic pulses. However, it is not known whether the same individuals were involved in these repeated observations in strongly ensonified areas, or whether individuals that tolerate repeat exposures

may still experience a stress response (see *Non-Auditory Physical or Physiological Effects* section below).

For sperm whales, available studies for a variety of anthropogenic sounds indicate these whales may or may not exhibit responses to such sounds, and that responses that do occur are variable. Based on available information, it appears sperm whales may react strongly to a novel acoustic stimulus but may habituate to the presence of some anthropogenic sounds (NMFS 2006b). Sperm whale responses to various anthropogenic sounds include disruptions of sperm whale clicking and behavior from sonars (Watkins and Schevill 1975; Watkins *et al.* 1985; Goold 1999), pingers (Watkins and Schevill 1975), the Heard Island Feasibility Test (Bowles *et al.* 1994), and the Acoustic Thermometry of Ocean Climate or ATOC (Calambokidis *et al.* 1998; Costa *et al.* 1998). Sperm whales have been observed to temporarily stop clicking in response to pinger sounds in the frequency range 6-13 kHz (Watkins and Schevill 1975); however, this response is thought to be one of listening, rather than of fear (NMFS 2006b). For example, sperm whales 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). Goold (1999) reported distinct changes in vocalizations of six sperm whales off Scotland during a shepherding operation by vessels driving the whales through a narrow channel using ship noise and echosounder/fishfinder emissions. A recent preliminary analysis of acoustic data from the northern Gulf of Mexico also indicates that sperm whales are, in some cases, affected by the passing of vessels, with fewer clicks and fewer whales detected afterwards (Ioup *et al*. 2005 *as cited in* NMFS 2006b). It is not known if this reflects a change in sound-producing behavior, or the physical movement of whales away from the source. Similar changes were also observed when the data were analyzed for the effects of a passing tropical storm (Newcomb *et al*. 2004 *as cited in* NMFS 2006b).

In contrast, other studies have shown a lack of response by sperm whales to anthropogenic sounds. Madsen and Møhl (2000) found that sperm whales did not alter their vocal activity when exposed to levels of 173 dB re 1 μPa (rms) from detonators. Sperm whales in the Mediterranean Sea were reported to continue calling when exposed to frequent and strong military sonar signals (J. Gordon pers. comm. *as cited in* Richardson *et al.* 1995). When André *et al.* (1997) exposed sperm whales to a variety of sounds to determine which sounds would scare whales away from paths of vessels, sperm whales were not observed to exhibit startle reactions to sources other than 10 kHz pulses (180 dB re 1 μPa at the source).

These studies demonstrate that sperm whales can be susceptible to certain anthropogenic sounds, though responses vary. As for a response by sperm whales to seismic surveys, limited systematic information is available regarding the reactions of any toothed whale to impulsive noises. However, information that is available indicates that for small and medium-sized toothed whales, the predominantly low-frequency seismic pulses (< 188 Hz) of the seismic airguns represent part of the spectrum where auditory systems are not very sensitive (i.e., higher hearing thresholds) (Richardson *et al.* 1995). Nonetheless, available information indicates seismic pulses are strong enough to be detectable to these small-to-moderate sized odontocetes, although avoidance reactions may be limited to considerably shorter ranges (Richardson and Würsig 1997; Goold and Fish 1998). In addition, reactions to impulse noise likely vary depending on the

activity at time of exposure – e.g., in the presence of abundant food or during sexual encounters toothed whales sometimes are extremely tolerant of noise pulses (NMFS 2006b).

Sperm whales are reported to show avoidance reactions to standard vessels not emitting airgun sounds (e.g., Richardson *et al.* 1995; Würsig *et al.* 1998), and it is reasonable to assume these whales may avoid an operating seismic survey vessel as well (L-DEO 2006). Accounts of possible avoidance of seismic vessels exist for sperm whales in the Gulf of Mexico (Mate *et al.* 1994; Jochens and Biggs 2004).

Some information indicates possible responses by sperm whales after exposure to seismic sources, such as an opportunistic observation by Mate *et al.* (1994), who reported a decrease in the number of sperm whales in a given area after the initiation of airgun seismic testing, and Johnson and Miller (2002 *as cited in* NMFS 2006g) who reported one tagged whale moving away from an operating seismic vessel in the northern Gulf of Mexico in July 2001 once received seismic pulses reached approximately 137 dB re 1 μPa. Sperm whales may also have responded to seismic airgun sounds by ceasing to call during some (but not all) times when seismic pulses were received from an airgun array more than 300 km away (Bowles *et al.* 1994).

Contrary to the observations mentioned above, results of other studies indicate there is considerable tolerance of seismic surveys by at least some sperm whales. Davis *et al.* (2000) noted that sighting frequency for sperm whales did not differ significantly between different acoustic levels used in the northern Gulf of Mexico. A recent study off northern Norway reported that sperm whales continued to call and remained in the area for at least 13 days when exposed to pulses from a distant seismic vessel, involving received levels up to 146 dB re 1 μPa pk-pk (i.e., 130 dB (rms) (Madsen *et al.* 2002). Similarly, a study conducted off Nova Scotia analyzing recordings of sperm whale sounds at various distances from an active seismic program did not detect any obvious changes in the distribution or behavior of sperm whales (McCall Howard 1999 *as cited in* NMFS 2006h). Recent data from vessel-based monitoring programs in the United Kingdom (U.K.) also suggested that sperm whales showed no noticeable avoidance response. Compilation and analysis of data on responses of marine mammals to seismic surveys off the U.K. did not result in statistically significant evidence of avoidance by sperm whales (Stone and Tasker 2006). One interpretation is that sperm whales have a high tolerance for certain types of noise (e.g., André *et al.* 1997).

An experimental study of sperm whale reactions to seismic surveys in the Gulf of Mexico was conducted, along with a study of the movements of sperm whales with satellite-linked tags in relation to seismic surveys (see Jochens and Biggs 2003; 2004; Jochens *et al.* 2006; Jochens 2008). Data show that during two controlled exposure experiments with exposure to seismic pulses at received levels up to 148 dB re 1 μPa (rms) over the octave band with most energy, there was no indication of avoidance of the vessel or changes in diving behavior (Jochens *et al.* 2006; Jochens 2008). In addition, Madsen *et al.* (2006) report that seven of eight tagged sperm whales continued to perform foraging dives throughout exposure to seismic airguns at levels up to 147 dB (rms) (the eighth whale remained resting at the surface during exposure). Although the sample sizes for these findings are small, the results are consistent with those off northern Norway by Madsen *et al.* (2002). Jochens *et al.* (2006) report that visual observations of sperm whale clusters during seismic studies in the Gulf indicated no significant responses in terms of

(1) heading relative to seismic surveys, (2) time spent at the surface during surveys, and (3) surfacing rate from two hours before and after seismic survey lines within 100, 50, or 25 miles. Although these studies suggest that sperm whales did not exhibit horizontal avoidance of seismic activity, few exposures occurred above 160 dB pk-pk (or approximately 144 dB rms), and further research is needed to examine avoidance at higher received levels (Jochens *et al.* 2006). Jochens *et al.* (2006) also speculate that sperm whales in that area may have some level of habituation to airgun sounds.

These studies suggest that sperm whales exhibit considerable tolerance of seismic sources (e.g., no apparent disruption of behaviors such as foraging or calling), or possibly some degree of habituation. Information on distance from airguns and received levels are not always provided in these studies; distance from airguns, which can determine the received level, has been found to be an important factor affecting other large whales species such as humpback (McCauley *et al.* 2000a; 2000b), gray, and bowhead whales (see Richardson *et al.* 1995).

For the ESA-listed whale species exposed to seismic airguns during the proposed activities, any alterations of normal behavior that result in avoidance of biologically important habitat or reductions in foraging opportunities could be biologically significant. The proposed activities would overlap with foraging and possibly migrating whales in the action area. If repeated displacement or disruption of animals occurred, the reproduction and recruitment rates could be reduced. However, as mentioned previously, the maximum repeat exposure of individuals expected during the proposed activities would be once, assuming whales do not alter their location between subsequent seismic firing of that transect. Although the proposed activities are expected to overlap with foraging whales, given the limited duration of the proposed seismic activities, the limited risk for repeat exposure to airguns and the mitigation measures to minimize the risk of exposure at received levels of concern, we do not anticipate that behavioral responses to the proposed activities would significantly result in reduced foraging opportunities. Any behavioral responses or disruptions in whale behavior resulting from exposure to seismic sounds are expected to be temporary, with whales expected to resume their behavior after the seismic vessel has moved out of their immediate area without impairment of feeding, migration, or other behaviors.

Behavioral Responses of Steller Sea lions

Visual monitoring from seismic vessels has shown slight (if any) avoidance of airguns by pinnipeds, and only slight (if any) changes in behavior. In the Beaufort Sea, some ringed seals avoided an area of 100 m to (at most) a few hundred meters around seismic vessels, but many seals remained within 100–200 m of the trackline as the operating airgun array passed by (e.g., Harris et al. 2001; Moulton and Lawson 2002; Miller et al. 2005). Ringed seal sightings averaged somewhat farther away from the seismic vessel when the airguns were operating than when they were not, but the difference was small (Moulton and Lawson 2002). Similarly, in Puget Sound, sighting distances for harbor seals and California sea lions tended to be larger when airguns were operating (Calambokidis and Osmek 1998). However, previous telemetry work suggests that avoidance and other behavioral reactions may be stronger than evident to date from visual studies (Thompson et al. 1998).

As for listed whale species above, any alterations of Steller sea lion normal behavior that results in avoidance of biologically important habitat, essential life functions or reductions in foraging opportunities could be biologically significant. The proposed activities would overlap with foraging and breeding Steller sea lions in the action area. If repeated displacement or disruption of animals occurred, the reproduction and recruitment rates could be reduced. However, as mentioned previously, the maximum repeat exposure of individuals expected during the proposed activities would be once, assuming sea lions do not alter their location between subsequent seismic firing of that transect. Although the proposed activities are expected to overlap with breeding and foraging sea lions, given the limited duration of the proposed seismic activities, the limited risk for repeat exposure to airguns and the mitigation measures to minimize the risk of exposure at received levels of concern, we do not anticipate that behavioral responses to the proposed activities would significantly result in reduced foraging or breeding opportunities. Even if sea lion responses upon exposure to acoustic sounds during the survey are as strong as those evident in the telemetry study mentioned above (Thompson et al. 1998), any behavioral responses or disruptions in Steller sea lion behavior are expected to be temporary, with sea lions expected to resume their behavior after the seismic vessel has moved out of their immediate area without impairment of feeding, breeding or other behaviors.

Behavioral Responses of Sea Turtles

Sea turtle hearing thresholds appear to be higher than those for mammals (DFO 2004; NMFS 2006h), and DFO (2004) concluded it is unlikely that sea turtles would be more sensitive to seismic operations than cetaceans, based on available studies. Sea turtles are expected to be less sensitive to sounds; however, behavioral responses to environmental sounds are documented in several controlled experiments. As mentioned previously, studies on sea turtle hearing indicate sensitivity to low frequency sounds (Ridgway *et al.* 1969; Lenhardt *et al.* 1983; Moein Bartol *et al.* 1999), and it has been suggested that sea turtles use acoustic signals from their environment as guideposts during migration and as a cue to identify their natal beaches (Lenhardt *et al.* 1983). Some possible reactions to low frequency sounds include startle responses and rapid swimming (Lenhardt 2002; McCauley 2001 *as cited in* NMFS 2006h), as well as swimming towards the surface at the onset of the sound (Lenhardt 1994).

Available studies suggest some sea turtles exhibit an avoidance reaction to airgun-generated sounds. McCauley *et al.* (2000a; 2000b) investigated the effects of airguns on sea turtle behavior. The authors found that green and loggerhead sea turtles show avoidance to airgun arrays at 2 km (1.1 nm) and at 1 km (0.54 nm) with received levels of 166 dB re 1 µPa and 175 dB re 1 µPa, respectively. Individual sea turtles responded consistently by noticeably increasing swimming activity above a level of approximately 166 dB re 1 µPa (rms), as compared to swimming during non-airgun operation periods. The increase in swimming behavior tracked the received airgun level, by increasing at increasing levels. Above 175 dB re 1 µPa, turtle behavior became more erratic, possibly indicating the turtles were in an agitated state. In studies by Lenhardt (1994) and Lenhardt *et al.* (1983), loggerhead and Kemp's ridley turtles responded to airgun pulses and low-frequency sound – e.g., by becoming active and swimming to the surface upon exposure. Moein *et al*. (1994) used an evoked potential method to test sea turtle hearing and reported avoidance behavior in loggerhead sea turtles at the beginning of airgun exposure trials; however, repeated airgun exposures days after the initial tests did not elicit a statistically significant avoidance response. The authors concluded this may be due to either habituation or

temporary threshold shift in the turtles hearing capability. And based on a review of sea turtle data from 11 L-DEO seismic surveys since 2003, Holst *et al*. (2006) concluded that turtles exhibited localized avoidance during both large- and small-source seismic surveys.

Although studies suggest sea turtles are most likely to avoid seismic airgun pulses, monitoring reports from seismic surveys indicate occasions when sea turtles were likely exposed to seismic airgun pulses – e.g., green, leatherback, and olive ridley turtles during surveys in the Hess Deep area of the eastern tropical Pacific Ocean (Smultea and Holst 2003 *as cited in* NMFS 2006h). Of six sea turtles that were sighted during those surveys, five were seen while airguns were active. In at least one instance, an olive ridley sea turtle was sighted within ten meters of the array while active. This turtle was reported to exhibit visible responses to either exposure to the seismic pulses or to the physical presence of the array and floats (NMFS 2006h). Although the turtle swam away from the vessel and was not reported to suffer physical injury, it is assumed the turtle experienced a stress response to its exposure that may have risen to the level of harassment. Similarly, Holst *et al.* (2005b) report during seismic operations in the southern Gulf of Mexico off the Yucatán Peninsula, that seven sea turtles were sighted within the 180 dB safety radius in shallow water (< 40 m). One of these turtles was reported to be actively swimming away from the seismic source. Six of these turtles were seen < 200 m from the operating airguns before the airguns were powered- or shut-down (Holst *et al.* 2005b).

In summary, available evidence indicates avoidance of seismic sources by sea turtles is likely, but that some turtles may not avoid the source vessel and may be exposed to seismic sound at levels of concern. Based on available information on captive turtles, avoidance may begin at received levels above 166 dB re 1 μPa. Avoidance behavior may shorten the exposure period, and the ramping-up of airguns during the proposed study would provide opportunity for avoidance by sea turtles, thereby minimizing exposure to received levels of concern. Avoidance or any disruptions in sea turtle behavior are expected to be temporary and are not expected to cause any injury. For those turtles that might be exposed to seismic pulses at levels above 166 dB re 1 μPa during the proposed study, we expect this could result in a stress response that rises to the level of harassment (see below) (NMFS 2006h). Based on the available information, we expect any turtles exposed to the seismic airguns would resume their behavior after the seismic vessel has moved out of the immediate area without significant impairment of foraging, migration, or other behaviors.

Non-Auditory Physical or Physiological Effects

Non-auditory physical or physiological effects are possible in marine mammals and turtles exposed to strong underwater pulsed sound, such as from airguns (LGL, Ltd 2011); however, studies of such effects are limited. Possible types of effects or injuries could include stress, neurological effects, and other types of organ or tissue damage (LGL, Ltd 2011).

Stress responses by animals involve the autonomic nervous system, producing changes in heart rate, blood pressure, and gastrointestinal activity that are typically of short duration. Such responses may or may not have significant long-term effects on an individual's welfare (NMFS 2006g). Stress responses may also involve the neuroendocrine system and hormones associated with the HPA-axis (hypothalamus-pituitary-adrenal system) in mammals or the hypothalamus-pituitary-interrenal axis in some reptiles. Functions affected by stress include immune

competence, reproduction, metabolism, and behavior (NMFS 2006g). Stress is an adaptive response and does not normally place an animal at risk; however, *distress* (allostatic loading) involves a stress response resulting in a biological consequence to the individual and lasts until the animal replenishes its energy reserve sufficient to restore normal function. Minimal information is available on the physiological responses of marine mammals and sea turtles upon exposure to anthropogenic sounds. Given studies of other marine and terrestrial animals, it would be reasonable to assume that some marine mammals might experience physiological stress responses that would be classified as "distress" upon exposure to mid- and low-frequency sounds. As whales use hearing as a primary way to gather information about their environment and for communication, we assume that limiting these abilities could be stressful for some individuals. Therefore, exposure to levels sufficient to trigger onset of PTS or TTS might be accompanied by physiological stress responses, as terrestrial animals are known to exhibit such responses under similar conditions (NRC 2003; NMFS 2006g). Stress responses may also occur at levels lower than those required for onset of TTS (NMFS 2006g). Although the magnitude and biological significance of any stress responses that might occur remain unknown, exposure to seismic sources would be limited in duration and some whales may exhibit some avoidance of seismic sources. Mitigation measures and monitoring are expected to help reduce the likelihood of exposure at levels of concern, further minimizing risk to fin and humpback whales and Steller sea lions.

It is possible that some marine mammal species may be susceptible to injury or stranding after exposure to seismic pulses; however there is no definitive evidence that these effects occur, even in close proximity to large airgun arrays (see discussion below under *Strandings*) (LGL, Ltd 2011). Available information indicates that gas and fat embolisms may potentially occur if cetaceans ascend too quickly when exposed to aversive sounds or if sounds in the environment cause the destabilization of existing bubble nuclei (see Potter 2004; Arbelo *et al.* 2005; Fernández *et al.* 2005a). There is speculation that gas and fat embolisms can occur during exposure to mid-frequency sonar; however, no available evidence indicates these effects occur in response to airgun sounds (LGL, Ltd 2011).

Sea turtles exposed to seismic sound may possibly experience a physiological stress response, but available studies are inconclusive (DFO 2004), and the magnitude or effects of any response remain largely unknown. Given evidence suggesting sea turtles likely avoid seismic sources, and those that are known to be exposed are not reported to have suffered a detectable physical injury, we assume that sea turtles exposed to seismic pulses during the proposed activities would not experience physical effects beyond a possible stress response. A stranding event involving sea turtles was reported coincident with seismic surveys (see *Strandings* section below); however, no available evidence definitively links seismic airgun testing with sea turtle mortality.

Although data are limited, we assume that some whales, Steller sea lions and Leatherback sea turtles may experience a stress response if exposed to seismic pulses in the proposed activities. Other non-auditory physical or physiological effects are unlikely to occur during the proposed activities given the limited duration of any exposures experienced by animals, the likelihood of at least some behavioral avoidance of seismic pulses, as well as efforts to minimize exposure via monitoring and mitigation measures. Such effects might only occur in unusual situations when

individuals are exposed at close range for unusually long periods of time (LGL, Ltd 2011); this is not anticipated to occur during the proposed activities.

Strandings

Available information indicates that marine mammals close to underwater detonations can be killed or severely injured, with auditory organs especially susceptible to injury (Ketten *et al.* 1993; Ketten 1995 *as cited in* LGL, Ltd 2008). However, seismic airgun pulses are less energetic and have slower rise times, and there is no evidence available conclusively linking airguns to serious injury, death, or stranding even in the case of large airgun arrays (Gordon *et al.* 2003; L-DEO 2006). Evidence implicating seismic airguns in the stranding of marine mammals does exist for two beaked whales in the Gulf of California (*Ziphius cavirostris* – not listed under the ESA nor currently considered a candidate for such listing), and possibly for four *Z. cavirostris* in the Galápagos Islands (Gentry 2002; Gordon *et al.* 2003). There appeared to be a temporal correlation between these events and seismic operations by the vessel *R/V Maurice Ewing* in the vicinity; however, a causal link could not be established for either event. Other strandings of beaked whales have also occurred, associated with military mid-frequency sonar transmissions – e.g., in the Bahamas, Canary Islands, and eastern Mediterranean Sea (Frantzis 1998). These other strandings were associated with military mid-frequency sonar (generally 2-10 kHz and relatively narrow bandwidth), which differs from the sound produced by seismic arrays (broadband and below 1 kHz). However, evidence that sonar pulses can lead to physical damage or mortality (even if indirectly) (U.S. Department of Commerce and U.S. Department of the Navy 2001; Jepson *et al.* 2003; Fernández *et al.* 2005) suggests that caution is still warranted when considering the effects on marine mammals from any high-intensity pulsed sound, such as from seismic airguns (LGL, Ltd 2008).

For leatherback sea turtles, no available information definitively links seismic airgun activities with any sea turtle mortalities (e.g., see 60 FR 21745; May 3, 1995) and strandings, although studies are limited. Anecdotal evidence from early 2004 indicates that more than 30 sea turtles stranded dead in Yucatán, Mexico (Jaszy and Horowitz 2005). Guzman-Hernandez (pers. comm. in NMFS 2006c) stated that one of the sea turtles had burst lungs, internal bleeding, and auditory damage. This event occurred during a time when seismic testing had been conducted nearby in very shallow waters. However, no definitive causal link is noted and seismic activity during the proposed study would be conducted mostly at deeper depths with monitoring to allow an opportunity to keep sea turtles from exposures to the highest received levels.

Given the available evidence on strandings, serious injury or mortality of sperm whales or sea turtles due to the proposed seismic activities is not anticipated. First, marine mammal strandings involved beaked whales, which exhibit a distinct ear anatomy compared to other cetaceans. In addition, published information suggests listed taxa would not be lethally affected by exposure to the proposed seismic surveys. Lastly, the mitigation measures to be used during the proposed seismic activities would help minimize or avoid exposure.

Effects on Prey

In addition to the responses discussed above, seismic surveys could have indirect, adverse effects on whales, Steller sea lions and leatherback sea turtles by reducing the abundance or availability of prey or changing the structure or composition of the fish community. These indirect effects

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 seismic surveys. Because fish and invertebrate species such as squid and jellyfish are pelagic prey for whales, Steller sea lions and leatherback sea turtles, such effects might have adverse consequences for individuals foraging in the action area.

Several studies have shown that short, sharp sounds can cause overt or subtle changes in fish behavior and distribution. Chapman and Hawkins (1969) tested the reactions of whiting (hake) in the field to an airgun. When the airgun was fired, the fish showed a sudden downward movement, changing their distribution from being dispersed between 25 m (80 ft) and 55 m (180 ft) depth, to forming a compact layer below 55 m (180 ft). Toward the end of an hour-long exposure to the airgun pulses the fish had habituated to the sound and risen back upward in the water column, despite the continued presence of sound pulses. However, when the airgun resumed firing after a dormant period, the fish exhibited another downward response. Pearson *et al.* (1992) conducted a controlled experiment to determine the effects of strong sound pulses on several species of rockfish off California. Exposing rockfish to an airgun with a source level of 223 dB re 1 μPa, the authors reported startle and alarm responses in these fish. Popper *et al.* (2005) report the occurrence of threshold shifts in some fish after exposure to airguns, with recovery in 24 hours. In other airgun experiments, catch-per-unit-effort (CPUE) of demersal fish was reported to decline when airgun pulses were emitted (Skalski *et al.* 1992; Dalen and Knutsen 1986, Dalen and Raknes 1985, both *as cited in* NMFS 2006h). Reductions in the catch may have resulted from a change in the behavior of the fish. Fish schools descended to near the bottom when the airgun was firing, and the fish may have changed their swimming and schooling behavior. Fish behavior returned to normal minutes after the sounds ceased. In the Barents Sea, abundance of cod and haddock measured acoustically was reduced by 44 percent within 9.2 km (5 nm) of an area where airguns operated (Engås *et al.* 1993 *as cited in* NMFS 2006h). Actual catches declined by 50 percent throughout the trial area and 70 percent within the shooting area. This reduction in catch decreased with increasing distance until 30-33 km (16-18 nm), where catches were unchanged.

McCauley *et al.* (2003) also conducted an experiment on the effects of airgun sounds on fish. Several fish were exposed to an operating airgun, over 1.5 hours at 10-second intervals with received levels varying from less than 100 dB to over 160 dB re 1 μPa. The exposure resulted in apparent permanent, extensive damage to their sensory epithelia. Although this study demonstrates fish can be injured from repeated exposure to airgun sounds, fish in the wild are likely to move away from a seismic source and are not expected to be exposed in such a manner. Other studies of the behavioral responses of fish and fishing success to seismic sources report similar responses – e.g., see Dalen and Knutsen (1986), Turnpenny and Nedwell (1994), LaBella *et al.* (1996), Kenchington (1999), Santulli *et al.* (1999), Hirst and Rodhouse (2000), Thomson *et al.* (2001), Wardle *et al.* (2001), and Engås and Løkkeborg (2002). Egg and larval stages would not be able to escape such exposures, and Gausland (2000) reported that seismic signal levels of 230-240 dB re 1μPa pk-pk (or 16 dB lower: 214-224 dB rms) are sufficient for harm to occur to fish eggs and larvae (see also Kostyuchenko 1973).

Limited information is available on the effects of seismic pulses on invertebrates such as squid or jellyfish. A range of invertebrates are reported to be sensitive to low-frequency (10–150 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 dominant frequency range of seismic pulses, indicating that invertebrates could likely perceive seismic 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*. (2000a; 2000b) 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 mean squared pressure. Alarm responses in squid were detected during gradual ramp-up of airguns once levels exceeded 156-161 dB re 1 μPa mean squared pressure. 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 2–5 km (1.1–2.7 nm) from an approaching large seismic source.

These studies indicate that seismic airgun activity has the potential to affect fish and invertebrates. Fish appear to exhibit startle responses and avoidance of seismic sources, recovering or habituating after a short time period. Squid also appear to exhibit alarm responses and avoidance of seismic sources. Limited data on the physiological effects of seismic sound on fish and invertebrates indicate these effects are short-term and most apparent after exposure at very close range. Disturbance of these prey species has the potential to negatively affect whale or sea turtle foraging in the action area. However, with the limited spatial and temporal scale of the proposed seismic activities, only a small fraction of available habitat would be ensonified at any one time, and prey species would be expected to return to their pre-exposure behavior once seismic firing ceased. Thus, we expect such responses would have only minor, temporary effects on the feeding ability of whales, Steller sea lions and leatherback sea turtles in the immediate survey area. Such reductions in feeding ability are not expected to reduce an individual animal's overall feeding success, and it does not appear likely that any effects on prey would pose significant risk to fin, humpback and sperm whales, Steller sea lions or leatherback sea turtles in the Action Area.

Effects of exposure to MBES and SBP

Sperm whales are presumed to be more sensitive to mid- and high-frequency sounds, and may be able to hear the mid-frequency sounds of the MBES and SBP sonar. Humpback whales and Steller sea lions may also detect sounds frm the MBES and SBP sonar. Any exposures to these sources would be brief as the vessel passes by and individual pulses will be very short. Potential for exposure is further reduced by the fact that sounds from these sources would dissipate over an area smaller than that affected by seismic airguns, for which mitigation measures would minimize exposure within the 180dB re 1μPa isopleths for whales and the 190dB re 1μPa isopleths for Steller sea lions.

In the unlikely event that a whale or sea lion is exposed to the sonar, these animals are likely to avoid the source. Sperm whales reacted to military sonar by dispersing from social aggregations,

moving away from the sound source, remaining relatively silent and becoming difficult to approach (Watkins *et al.* 1985). Experiments on captive odontocetes provide additional information. Captive bottlenose dolphins and a beluga whale exhibited changes in behavior when exposed to 1-second pulsed sounds at frequencies similar to those emitted by multi-beam sonar (Ridgway *et al.* 1997; Schlundt *et al.* 2000), and to shorter broadband pulsed signals (Finneran *et al.* 2000; 2002). Behavioral changes typically involved apparent attempts at avoidance (of the sound exposure, itself, or the location of the exposure site during subsequent tests) (Finneran *et al.* 2000; Schlundt *et al.* 2000). Dolphins exposed to 1-second intense tones exhibited short-term changes in behavior above received sound levels of 178 to 193 dB re 1 μPa (rms), as did belugas at received levels of 180 to 196 dB re 1 μPa and above. For shorter pulses, received levels necessary to elicit such reactions were higher (Finneran *et al.* 2000; 2002). Test animals sometimes vocalized after exposure to pulsed, mid-frequency sound from a watergun (Finneran *et al.* 2002), and in some instances animals exhibited aggressive behavior toward the test apparatus (Ridgway *et al.* 1997; Schlundt *et al.* 2000). The relevance of these data to free-ranging odontocetes is uncertain. In the wild, cetaceans sometimes avoid sound sources before they are exposed to the levels listed above, and reactions in the wild may be more subtle than those described by Ridgway *et al.* (1997) and Schlundt *et al.* (2000).

Recent stranding events associated with the operation of naval sonar suggest that mid-frequency sonar sounds may have the capacity to cause serious impacts to marine mammals (see *Strandings* above). However, the sonars proposed for use by USGS differ from sonars used during Naval operations, which generally have a longer pulse duration and are often directed close to horizontal as opposed to the more downward-directed MBES and SBP. The sound energy that would be received by any individuals exposed to the MBES and SBP sources during the proposed activities is lower relative to naval sonars (LGL Ltd. 2011), as is the duration of exposure. In addition, the area of possible influence for the MBES and SBP are much smaller, consisting of a narrow zone close to and below the source vessel. Because of the unlikelihood of exposure and the brief duration for any individual that might be exposed, it is not likely MBES and SBP sonar pose a risk to sperm or humpback whales or Steller sea lions during the proposed activities. In addition, Boebel *et al.* (2006) assessed the relative risk posed by various scientific acoustic instruments and concluded that multi-beam systems and sub-bottom profilers similar to those to be used during the proposed activities presented a low risk for auditory or any other injuries, and that an individual would require exposure to 250–1,000 pulses from a sub-bottom profiler to be at risk for TTS. To be susceptible to TTS, a whale or sea lion would have to pass at very close range and match the vessel's speed – the probability of this occurring in the proposed survey is expected to be very small. Masking of fin, humpback or sperm whale or Steller sea lion communications is not expected to occur appreciably due to MBES or SBP signals given their directionality, their low duty cycle, and the brief period when an individual mammal is likely to be within its beam (LGL Ltd. 2011).

For leatherback sea turtles and fin whales, available information indicates detection of sounds in the low-frequency range; based on this information, any individuals of these species exposed to received levels of mid-frequency (1 kHz–10 kHz) sounds or higher are not likely to detect these sounds. Therefore, fin whale and leatherback sea turtles are not likely to respond physiologically or behaviorally to received sounds from the MBES and SBP sonar to be used by the *Langseth.*

Effects of exposure to OBS
The acoustic release transponder used to communicate with the OBSs uses frequencies of 9–13 kHz. Humpback and sperm whales could hear these signals, although fin whales and leatherback sea turtles would not. Because these signals will be used very intermittently throughout the 2 days of recovery, it is unlikely that humpback and sperm whales and Steller sea lions would respond to these signals. Since these frequencies are not audible to sea turtles, we do not expect them to hear and, therefore, respond to the signals.

Cumulative Effects

Cumulative effects include the effects of future state, tribal, local or private actions that are reasonably certain to occur in the action area considered by this Opinion. Future federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the ESA.

During this consultation, NMFS searched for information on future state, tribal, local, or private actions reasonable certain to occur in the action area. NMFS expects the natural phenomena in the action area (e.g., climate change, natural mortality) will continue to influence listed species as described in the *Environmental Baseline*.

We also expect anthropogenic effects described in the *Environmental Baseline* will continue, including those related to habitat degradation due to pollution, contaminants and ocean noise, vessel traffic; and commercial fishing. An increase in these activities could result in an increased effect on ESA-listed species. However, the magnitude and significance of any anticipated effects remain unknown at this time.

Integration and Synthesis of Effects

The United States Geological Survey proposes to conduct a marine geophysical survey on board the *R/V Langseth* in the central Gulf of Alaska. NMFS' Office of Protected Resources Permits, Conservation and Education Division proposes to issue an IHA for incidental takes that would occur during this survey, pursuant to MMPA section 101(a)(5)(D).

As explained in the *Approach to the Assessment* section, risks to listed individuals are measured using changes to an individual's "fitness" – i.e., the individual's growth, survival, annual reproductive success, and lifetime reproductive success. When listed plants or animals exposed to an action's effects are not expected to experience reductions in fitness, we would not expect the action to have adverse consequences on the viability of the population(s) those individuals represent or the species those populations comprise (Brandon 1978; Mills and Beatty 1979; Stearns 1992; Anderson 2000). As a result, if the assessment indicates listed plants or animals are not likely to experience reductions in their fitness, we conclude our assessment. If possible reductions in individuals' fitness are likely to occur, the assessment considers the risk posed to population(s) and then to the species to which those individuals belong.

It is important to note that the studies available to inform our risk assessment are limited, including information on the effects of anthropogenic noise (i.e., seismic pulses and bathymetric sonar) on listed whales, pinnipeds and sea turtles. Information on these effects is limited and

some methods to acquire acoustic information are not available. Underwater hearing abilities have been studied in a few species, and where experimental data do not exist we have made inferences based on the characteristics of sounds or from hearing physiology. Definitive statements on the effects of sound are complicated because detection of sound by species depends on acoustic properties, transmission characteristics, and hearing sensitivity. In addition, responses by an individual animal can be highly variable and depend on its activity at the time of exposure, its age, any habituation to sounds, and other factors. The narratives that follow integrate and synthesize the information contained in the *Status of the Species*, the *Environmental Baseline*, and the *Effects of the Action* sections of this Opinion to assess the risk the proposed activities pose to fin, humpback and sperm whales, Steller sea lions and leatherback sea turtles. We are not aware of any cumulative effects (i.e., from future state, local, tribal, or private actions) that would alter our risk assessment for these species.

Whales

Fin whales, humpback whales and sperm whales are all endangered throughout their ranges, with the primary cause for depletion being historic commercial whaling of these species. Although most commercial whaling has ceased, Japan still harvests sperm whales under an IWC scientific reseach permit.

Available abundance estimates for these whales species indicate that current abundance appears to be significantly lower than historic levels in all ocean basins, and whales remain rare in some formerly important habitats. Exceptions include possible increases in humpback whale abundance in the North Atlantic and portions of the Southern Hemisphere.

In the action area, fin whales, humpback whales and sperm whales may occur during the proposed activities. Hearing in sperm whales is thought to include mid and high frequencies, and hearing in fin whales is thought to include low frequencies. Humpback whales also are thought to exhibit low-frequency hearing, with sensitivity extending into the mid-frequency range.

At present, there are several factors that may be affecting whale survival and recovery in the action area, although the significance of any effects remains largely unknown. Natural factors include circulation and productivity patterns affecting prey distribution and habitat quality; as well as natural mortality of whales, which we assume includes predation biotoxins, parasites, and disease. Anthropogenic factors include degradation of habitat resulting from pollution/ contaminants, anthropogenic noise, risk of ship strikes and entanglement or entrapment in fishing gear. Conservation and management efforts are also ongoing but any positive effect on whales in the action area will hopefully incease in the future.

After reviewing the available information, the proposed activities are likely to produce two potential stressors for listed whales: (1) acoustic energy from the airgun array and (2) acoustic energy from the MBES and SBP sonars. We believe an individual whale would have a low probability of being exposed to acoustic energy produced by the seismic airguns that will be used during the proposed action at received levels above 180 dB re 1 μPa. Using the model prepared by L-DEO (LGL Ltd. 2011), which estimates propagation distances for given received levels, seismic airgun levels of 180 dB re 1 μPa or greater might propagate up to 940 m in the survey

area for the 36 airgun array and 40 m for the 40 in^3 single airgun. These represent the maximum estimated distances for received levels (rms), and all of these distances fall within the visibility (i.e., monitoring) range from the *Langseth*. Although this consultation is primarily concerned with exposure to sound levels above 180 dB re 1 μPa, the estimates for listed whale exposure consider received levels at or greater than 160 dB re 1 μPa. Using the total area that would be ensonified ≥ 160 dB during the proposed activities, it is estimated that 76 fin whales, 68 humpback whales 10 sperm whales might be exposed to seismic pulses at these levels, with individuals exposed a maximum of once. These estimates do not account for possible avoidance of seismic sounds by whales, or for mitigation measures to be used during the studies that would reduce the risk of exposure to levels above 180 dB re 1 μPa. Although seismic pulses may propagate substantial distances beyond the isopleth for 160 dB re 1 μPa, we do not expect incidental harassment of listed species at those lower received levels.

Exposure of listed whales to the MBES and SBP sonars is expected to be minimal, given the brief ping duration, the limited area ensonified compared the the airgun, and the fact that the vessel will be in transit. The probability of a whale swimming through an area of exposure to MBES or SBP sonar is considered small. Any exposures that may occur are expected to be brief, and individuals are likely to exhibit avoidance. The potential for exposure to MBES and SBP sources is further reduced by the mitigation measures for minimizing exposure to seismic airguns within the 180 dB radius. Although recent stranding events involving beaked whales have been associated with the operation of naval mid-frequency sonar, the characteristics of the MBES and SBP sonars are significantly different, including a shorter pulse duration, general downward-orientation, significantly less sound energy that would be received, shorter exposure, and much smaller zone of influence close to the vessel. The probability of TTS occurring is considered very small, and masking is not expected to occur due to the short pulse duration and low likelihood of exposure. It is not likely that the use of MBES and SBP sonars during the proposed activities poses a significant risk to fin, humpback or sperm whales.

Although we cannot estimate the age, gender, or reproductive condition of any fin or humpback whales that might be exposed to these potential stressors, we assume these whales could represent any age class or either gender. Sperm whales in the Action Area would be juvenile and adult males. Based on the best available abundance information and population designations, whales exposed would represent a very small fraction of the regional abundance in the action area. The 76 fin whales would represent 0.47 percent of the regional abundance. The 68 humpback whales and 10 sperm whales would represent 0.33 percent, and 0.04 percent respectively, of their regional populations.

Possible effects of exposure to stressors described above could include hearing threshold shifts (TTS, PTS), masking or auditory interference, behavioral responses, or non-auditory physical or physiological effects. There is no evidence that exposures to airgun pulses can cause PTS, and we do not expect PTS to occur. Available data indicate that TTS is unlikely to occur unless whales are exposed to levels over 180 dB re 1 μPa (rms). Given the measures in the IHA, any exposures that may occur are more likely to be less than 180 dB re 1 μPa, and we consider TTS unlikely to occur as a result of the proposed survey. If an individual were to experience TTS, it is expected to be temporary and reversible, and even if repeated is not expected to cause permanent auditory damage.

The proposed seismic activities might mask whale calls at some of the lower frequencies, which could affect communication or echolocation, particularly for the baleen whales. However, for any frequencies that may overlap, given the low duty cycle of airguns and the fact that the source vessel will be in transit, it is not expected that exposure to seismic activities would pose a risk to individual whales due to masking.

Exposure to seismic activities may result in behavioral responses by listed whales, but responses are variable. Available information indicates that baleen whales generally tend to avoid operating airguns, with variable avoidance distances. However, some individual humpback whales are noted to approach airguns to distances where the received level was 179 dB re 1μPa (rms). Some sperm whales appear to tolerate seismic sound. Sperm whales are also reported to avoid standard, non-seismic vessels, and we expect individuals may also show some avoidance of seismic vessels, given accounts of possible avoidance of seismic vessels in other locations. A few observations indicate possible behavioral responses including avoidance or cessation of calling by sperm whales. However, studies indicate considerable tolerance of seismic activity, with whales continuing to call and maintaining their distribution and pre-exposure behavior. This suggests variable responses, with some sperm whales exhibiting considerable tolerance and others avoidance behavior. The proposed activities would occur during the breeding season for humpback whales; however, given the limited duration of the proposed activities at each leg in the surveys, the limited risk of repeat exposure to airguns and the mitigation measures to minimize risk of exposure at sound levels of concern, we do not expect the proposed activities would result in reduced foraging opportunities. Any behavioral disturbance of fin, humpback, or sperm whales that might occur is expected to be temporary, with whales expected to resume their behavior after the seismic vessel has moved out of the immediate area.

Stress responses may occur as a result of exposure, given the importance of sound and hearing to listed whales. Although the magnitude and biological significance of any stress responses that might occur remain unknown, we assume that stress responses would be minimized because exposure to seismic sources would be limited in duration and whales may show some avoidance of seismic sources. In addition, mitigation measures and monitoring would help minimize the risk to listed whales. Other non-auditory physical or physiological effects are considered unlikely to occur, given the limited duration of any exposure and the possibility of at least some avoidance of seismic pulses. Although two stranding events involving beaked whales are known to have occurred concurrent with seismic surveys (Gentry 2002; Gordon *et al.* 2003), no causal link is established for those events and no such records exist for listed whale species. The low occurrence of strandings concurrent with seismic studies is important to note, given the substantial amount of seismic activity that occurs in the marine environment. Underwater detonations are known to cause physical injury; however, seismic pulses are less energetic and have slower rise times, and no evidence links airguns to serious injury, death, or strandings of these whales.

Indirect effects on fin, humpback and sperm whales from exposure to seismic pulses are not likely to be significant. Effects from seismic airguns on the main prey items for whales would be short-term and affect a small fraction of available habitat and prey. Any prey that would be exposed are expected to recover quickly after exposure.

In summary, we do not expect injury or mortality to result from the proposed actions, and mitigation measures would help avoid exposure of whales at higher received levels. We expect that any individuals exposed to the proposed activities may be incidentally harassed, and as a result experience stress responses or exhibit behavioral responses to that exposure. The evidence available leads us to conclude that exposure to seismic pulse energy from the proposed seismic activities is not likely to cause a reduction in an individual whale's growth, survival, annual reproductive success, or lifetime reproductive success (i.e., fitness). As a result, we do not expect the proposed action to have an effect on the extinction risk of the population(s) these individuals represent or the whale species these population(s) comprise.

Steller Sea Lions

The Eastern Steller sea lion DPS is threatened through its range while the Western DPS is endangered throughout its range. The primary cause for depletion was the historic commercial hunting of both DPSs. Available abundance estimates indicate that the eastern DPS appears to be growing such that it may be a candidate for removal from the list of threatened and endangered species. Despite incomplete surveys in 2006 and 2007, the available data indicates that the western DPS has been stable since 2004.

At present, there are several factors that may be affecting Steller sea lion survival and recovery in the Action Area. Natural factors include circulation and productivity patterns affecting prey distribution and habitat quality; as well as natural mortality, which we assume includes predation, parasites, diseases, and exposure to biotoxins. Anthropogenic factors include degradation of habitat resulting from pollution/contaminants, anthropogenic noise, and entanglement or entrapment in fishing gear.

In the action area Steller sea lions may occur during the proposed activities. Hearing in Steller sea lions is believed sto pan low to mid-frequencies. After reviewing the available information, the proposed activities are likely to produce two potential stressors for listed Steller sea lions: (1) acoustic energy from the airgun array and (2) acoustic energy from the MBES and SBP sonars.

Steller sea lions are likely to be exposed to received levels greater than 170 dB re 1 µPa (rms), produced by acoustic signals from survey activities. This received level would constitute harassment for Steller sea lions. The maximum distance from airguns where received levels might meet the shut-down criterion of 190 dB (i.e., from the full 36-gun array and the single airgun) are estimated as 0.40 km for the 36 airgun array and 0.012 km for the 40 in^3 single airgun. Received levels above the 190 dB radius (the shut-down criterion) will not always reach these distances, as shorter radii will occur during the use of smaller numbers of airguns (e.g., the use of a single airgun during turns or power-down procedures).

These maximum distances all fall well within the visibility range from the *Langseth.* When stationed on the observation platform on the *Langseth*, eye level is about 21.5 m (70.5 ft) above sea level and PSVOs would be able to see around the entire vessel and to a distance of about 10 km with the naked eye, 5 km or further with the Big Eyes, 200 m with night vision and 2-3 km or further with the reticle binoculars. However, based on these maximum propagation distances,

our concern is the probability of ESA-listed whales, Steller sea lions and leatherback sea turtles occurring within this range from the *R/V Langseth* during seismic operations.

The best estimates for the number of Steller sea lions exposed to received levels greater than 170 dB re 1 μPa (rms) is 256 animals representing 0.61 percent of the regional population. Steller sea lions likely to be exposed to received levels greater than 170 dB re 1 μPa (rms), would be male and female juveniles and adults from either the Eastern or Western DPS. As Steller sea lions can breed as early as two weeks after giving birth, we expect that breeding animals would also be present during survey activities. Mitigation measures and monitoring activities during the proposed survey include visual and passive acoustic monitoring, an exclusion zone within the 180 dB isopleths for cetaceans and leatherback sea turtles, an exclusion zone within the 190 dB isopleths for Steller sea lions, speed and course alterations when practicable, power-down and shut-down procedures, an emergency shut-down provision, and ramp-up procedures for airguns. These measures are expected to reduce the risk that Steller sea lions would occur within the 190 dB radius; therefore, any exposures that might occur are more likely to involve Steller sea lions at received levels less than 190 dB re 1 μPa.

Based on the hearing abilities of Steller sea lions we would expect that sound pulses from the multi-beam and sub-bottom sonar would be audible to individuals of this species within the narrow extent of a transmitted sound beam. However, Kremser *et al.* (2005) concluded the probability of a cetacean swimming through the area of exposure when such sources emit a pulse is small, as the animal would have to pass at close range and be swimming at speeds similar to the vessel to be subjected to sound levels that could cause temporary threshold shifts. Similarly for Steller sea lions, we would expect the probability of swimming through the area of exposure when such sources emit a pulse is also small.

Base on the hearing capabilities of cetaceans and pinnipeds and the frequencies of the sound pulses from the OBS, we expect that Steller sea lions would hear the acoustic release transponder and the response but because these signals will be used very intermittently throughout the 2 days of recovery, it is unlikely that humpback and sperm whales and Steller sea lions would respond to these signals.

In pinnipeds, TTS thresholds associated with exposure to brief pulses (single or multiple) of underwater sound have not been measured. Initial evidence from more prolonged (non-pulse) exposures suggested that some pinnipeds (harbor seals in particular) incur TTS at somewhat lower received levels than do small odontocetes exposed for similar durations (Kastak et al. 1999, 2005; Ketten et al. 2001). The TTS threshold for pulsed sounds has been indirectly estimated as being an SEL of ~171 dB re 1 μPa^2 s (Southall et al. 2007), which would be equivalent to a single pulse with received level ~181–186 dB re 1 μPa(rms), or a series of pulses for which the highest rms values are a few dB lower. Mitigation measures mentioned above for whales would also apply to Steller sea lions and would help reduce the received level of any exposures that may occur and further minimize the risk of PTS or TTS.

Interference, or masking, generally occurs when the interfering noise is of a similar frequency and louder than the auditory signal received by an animal processing echolocation signals or listening for acoustic information from other individuals. Male Steller sea lions usually produce

low frequency roars (Loughlin et al. 1987; Schusterman et al. 1970). The calls of females range from 30 Hz to 3 kHz, with peak frequencies from 150 Hz to 1 kHz; typical duration is 1.0 to 1.5 sec (Campbell et al. 2002). Although the frequencies of seismic pulses overlap the range of Steller sea lion communication, the strongest spectrum levels of airguns are below 200 Hz (0-188 Hz for the *Langseth* airguns). The proposed seismic survey could mask some Steller sea lion calls, however, any masking that might occur would likely be temporary because seismic sources are discontinuous and the seismic vessel would continue to transit.

Visual monitoring from seismic vessels has shown slight (if any) avoidance of airguns by pinnipeds, and only slight (if any) changes in behavior. However, previous telemetry work suggests that avoidance and other behavioral reactions may be stronger than evident to date from visual studies (Thompson et al. 1998). Although the proposed activities are expected to overlap with breeding and foraging sea lions, given the limited duration of the proposed seismic activities, the limited risk for repeat exposure to airguns and the mitigation measures to minimize the risk of exposure at received levels of concern, we do not anticipate that behavioral responses to the proposed activities would significantly result in reduced foraging or breeding opportunities. Even if sea lion responses upon exposure to acoustic sounds during the survey are as strong as those evident in the telemetry study mentioned above (Thompson et al. 1998), any behavioral responses or disruptions in Steller sea lion behavior are expected to be temporary, with sea lions expected to resume their behavior after the seismic vessel has moved out of their immediate area without impairment of feeding, breeding or other behaviors.

Although data are limited, we assume that some Steller sea lions and may experience a stress response if exposed to seismic pulses in the proposed activities. Other non-auditory physical or physiological effects are unlikely to occur during the proposed activities given the limited duration of any exposures experienced by animals, the likelihood of at least some behavioral avoidance of seismic pulses, as well as efforts to minimize exposure via monitoring and mitigation measures. Such effects might only occur in unusual situations when individuals are exposed at close range for unusually long periods of time (LGL, Ltd 2011); this is not anticipated to occur during the proposed activities.

Seismic surveys could have indirect, adverse effects on Steller sea lions through reductions in the abundance or availability of prey or changing the structure or composition of the fish community. If prey fish experience lethal or sub-lethal damage, stress responses, or alterations in their behavior or distribution in response to acoustic energy produced by seismic surveys there may be adverse consequences for Steller sea lions foraging in the Action Area.

Studies indicate that fish appear to exhibit startle responses and avoidance of seismic sources, recovering or habituating after a short time period. Limited data on the physiological effects of seismic sound on fish indicate these effects are short-term and most apparent after exposure at very close range. With the limited spatial and temporal scale of the proposed seismic activities, only a small fraction of available habitat would be ensonified at any one time, and prey species would be expected to return to their pre-exposure behavior once seismic firing ceased. We expect such responses would have only minor, temporary effects on the feeding ability of Steller sea lions in the immediate survey area. Such reductions in feeding ability are not expected to reduce an individual animal's overall feeding success.

In summary, we do not expect injury or mortality to result from the proposed actions, and mitigation measures would help avoid exposure of sea lions at higher received levels. We expect that any individuals exposed to the proposed activities may be incidentally harassed, and as a result experience stress responses or exhibit behavioral responses to that exposure. The evidence available leads us to conclude that exposure to seismic pulse energy from the proposed seismic activities is not likely to cause a reduction in an individual sea lion's growth, survival, annual reproductive success, or lifetime reproductive success (i.e., fitness). As a result, we do not expect the proposed action to have an effect on the extinction risk of the population(s) these individuals represent or the species these population(s) comprise.

Sea Turtles

The primary causes for depletion of leatherback sea turtles were overexploitation and incidental capture in fishing gear. Reliable estimates of overall historic and current abundance for leatherback sea turtles are not available; however, information on nesting trends is available from many locations. Declines in abundance are reported rangewide for leatherback sea turtles; however, a few exceptions include increases reported for leatherback nesting in some locations of the Atlantic. The population structure of sea turtles is complex both spatially and genetically, with individuals exhibiting natal homing to nesting beaches and mixing of nesting aggregations on foraging grounds.

At present, there are several factors that may be affecting sea turtle survival and recovery in the action area. Natural factors include circulation and productivity patterns affecting prey distribution and habitat quality; as well as natural mortality of sea turtles, which we assume includes predation, parasites, diseases, and exposure to biotoxins. Anthropogenic factors include degradation of habitat resulting from pollution/contaminants, anthropogenic noise, and entanglement or entrapment in fishing gear.

After reviewing the available information, the proposed activities would produce the same two potential stressors for sea turtles, as noted above for listed whales: (1) acoustic energy from the airgun array and (2) acoustic energy from the MBES and SBP sonars. However, given that sea turtles are expected to detect and respond to sounds in the low-frequency range, any exposure to the mid-frequency or higher sounds such as MBES and SBP sonars is not likely to generate a response in leatherback sea turtles during the proposed activities.

Hearing in sea turtles is thought to include low frequencies, therefore we expect leatherback sea turtles to hear the acoustic signals from firing airguns. We expect sea turtles may be present during the proposed activities and that some may be exposed to received levels at or above 160 dB re 1 μPa. Similar to listed whales, mitigation measures to be used during the studies are expected to reduce the risk of sea turtle exposure to levels above 180 dB re 1 μPa, and we expect exposures that might occur for sea turtles would more likely involve received levels less than 180 dB re 1 μPa. Although seismic pulses may propagate substantial distances beyond the isopleth for 160 dB re 1 μPa, we do not expect incidental harassment of listed sea turtles at received levels below 166 dB re 1 μPa. Given the sparsity of sea turtles abundance information for the Gulf of Alaska we cannot estimate how many sea turtles may be exposed to airgun noise at received levels $\geq$ 160 dB during the proposed activities. We assume individual leatherback

sea turtles in the Action Area would represent either sex and any marine stage. Female leatherback sea turtles would be members of either the Costa Rican or Mexican nesting population. We cannot estimate the proportions of individual female leatherback sea turtles in the Action Area belonging to each population. We cannot estimate the population affiliation of male leatherback sea turtles that might be exposed since males from other nesting aggreagations tend to mix on the foraging grounds.

Possible effects on sea turtles from exposure to the seismic airguns could include hearing threshold shifts (TTS, PTS), masking or auditory interference, behavioral responses, or non-auditory physical or physiological effects. Available information indicates that TTS may occur in sea turtles exposed to seismic sources, although the received levels that may have elicited TTS are not known and experiments involved confined turtles. Sea turtles at sea would be expected to exhibit some avoidance behavior, and thus reduced exposure to seismic pulses. In addition, mitigation measures and monitoring are expected to reduce the risk of exposure at higher received levels. Given that sea turtle hearing capabilities are centered in the low-frequency range, exposure to the low-frequency seismic pulses may lead to masking. However, the seismic pulse would last a fraction of a second at intervals of 22 seconds or 66 seconds, thus any masking that may occur would be temporary and is not likely to present a significant risk for sea turtles that may be exposed.

Exposure may result in behavioral responses by sea turtles, since studies indicate some turtles exhibit avoidance reactions to airgun noises. Some accounts suggest sea turtles have been exposed to seismic sound resulting in a stress response that may have risen to the level of harassment. No injury or mortality of sea turtles is expected; the lack of strandings associated with seismic studies is important to note, particularly with the substantial amount of seismic activity in the marine environment. Evidence suggests that sea turtles exhibit behavioral responses, but no sea turtles have been reported to have suffered detectable physical injuries because of these exposures. Also, based on the evidence, sea turtles are likely to avoid seismic sources at harmful levels. Given this, we expect any exposed sea turtles would experience physical effects such as behavioral responses and possible stress responses that rise to the level of harassment, but no sea turtles are expected to be injured or killed from exposure to seismic sources.

The proposed activities are anticipated to occur during the leatherback foraging activities, however, exposure to airgun sounds are not expected to reduce foraging opportunities to levels that would reduce the fitness of individual leatherback sea turtles.

Indirect effects are not likely for leatherback sea turtles resulting from effects of airguns on prey. Any effects from airguns on prey would be short-term and affect a small fraction of available habitat and prey. Any prey that would be exposed are expected to recover quickly after exposure.

In summary, TTS and PTS are considered not likely to occur as a result of the proposed activities, because of avoidance behavior of other sea turtle species and reduced exposure risk at higher received levels resulting from the mitigation measures and monitoring. Masking is not expected to pose a significant risk to sea turtles. We do not expect injury or mortality. We

expect that individuals exposed to the proposed activities may be incidentally harassed, and as a result experience stress responses or exhibit behavioral responses to exposure. However, available information indicates some sea turtles would likely avoid seismic pulses. In addition, any exposures that may occur would be of short duration. The evidence available leads us to conclude that exposure to seismic pulse energy in the proposed survey is not likely to cause a reduction in an individual turtle's growth, survival, annual reproductive success, or lifetime reproductive success (i.e., fitness). As a result, we do not expect the proposed action to have an effect on the extinction risk of the populations these individuals represent or the sea turtle species those populations comprise.

Conclusion

After reviewing the current status of blue, fin, sei, humpback, North Pacific right and sperm whales, Steller sea lions, Pacific salmon and trout, and leatherback, green, olive ridley, and loggerhead sea turtles; the environmental baseline for the action area; the anticipated effects of the proposed activities; and the cumulative effects, it is NMFS' biological opinion that the proposed actions, as described in this Opinion, are not likely to jeopardize the continued existence of these species. Similarly, it is NMFS' biological opinion that the issuance of an IHA by NMFS' Office of Protected Resources Permits, Conservation and Education Division for harassment that would occur incidental to the proposed action is not likely to jeopardize the continued existence of these species. Critical habitat will not be adversely affected by the proposed actions.

INCIDENTAL TAKE STATEMENT

Section 9 of the ESA and federal regulation pursuant to section 4(d) of the ESA prohibit the "take" of endangered and threatened species, respectively, without special exemption. "Take" is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct. Harm is further defined by NMFS to include significant habitat modification or degradation that results in death or injury to listed species by significantly impairing essential behavioral patterns, including breeding, feeding, or sheltering. Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. Under the terms of sections 7(b)(4) and 7(o)(2), taking that is incidental and not intended as part of the agency action is not considered to be prohibited taking under the ESA provided that such taking is in compliance with the terms and conditions of this Incidental Take Statement.

The measures described below are nondiscretionary, and must be undertaken by the United States Geological Survey and the NMFS' Office of Protected Resources Permits, Conservation and Education Division 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 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.

Section 7(b)(4)(C) of the ESA specifies that in order to provide an incidental take statement for an endangered or threatened species of marine mammal, the taking must be authorized under section 101(a)(5) of the MMPA. One of the federal actions considered in this Opinion is the NMFS' Permits, Conservation and Education Division's proposed authorization of the incidental taking of fin, humpback and sperm whales pursuant to section 101(a)(5)(D) of the Marine Mammal Protection Act. With this authorization, the incidental take of listed whales is exempt from the taking prohibition of section 9(a), pursuant to section 7(o) of the ESA.

NMFS anticipates the incidental harassment of the fin whales (*Balaenoptera physalus*), humpback whales (*Megaptera novaeangliae*), and sperm whales (*Physeter macrocephalus*), Steller sea lions (*Eumetopias jubatus*) as well as leatherback sea turtles (*Dermochelys coriacea*) during the proposed survey activities.

Amount or Extent of Take

NMFS anticipates the proposed action to conduct a seismic survey in the central Gulf of Alaska might result in the incidental take of listed species. Fin, humpback, and sperm whales as well as leatherback sea turtles may be exposed to seismic sounds at received levels above 160 dB re 1 μPa. The proposed action might take 76 fin whales, 68 humpback whales and 10 sperm whales by exposing individuals to received levels greater than 160 dB re 1 μPa. These estimates are

based on the best available information on whale densities in the area to be ensonified above 160 dB re 1 μPa during the proposed activities. This incidental take would result from exposure to acoustic energy during seismic operations, would be in the form of harassment, and is not expected to result in the death or injury of any individuals that are exposed.

We anticipate the proposed action might result in the incidental take of Steller sea lions. Steller sea lions may be exposed to seismic sounds at received levels above 170 dB re 1 μPa. The proposed action might take 256 Steller sea lions by exposing individuals to received levels greater than 170 dB re 1 μPa. These estimates are based on the best available information on Steller sea lion densities in the area to be ensonified above 170 dB re 1 μPa during the proposed activities. This incidental take would result from exposure to acoustic energy during seismic operations, would be in the form of harassment, and is not expected to result in the death or injury of any individuals that are exposed.

We also expect the proposed action might also take individual leatherback sea turtles as a result of exposure to acoustic energy during seismic studies, and we expect this take would also be in the form of harassment, with no death or injury expected for individuals exposed. Harassment of leatherback sea turtles is expected to occur at received levels above 166 dB re 1 μPa. Because density estimates of sea turtles in the survey area are unknown, we estimate take as the number of turtles exposed to seismic operations above 166 dB re 1 μPa during the proposed activities. These turtles could be of all ages and life stages in the survey area.

Harassment of fin, humpback and sperm whales exposed to seismic studies at levels less than 160 dB re 1 μPa, of Steller sea lions at levels less than 170 dB re 1 μPa, or of sea turtles at levels less than 166 dB re 1 μPa, is not expected. We do not expect listed species to be taken by operation of the sonars. However, if overt adverse reactions (for example, startle responses, dive reactions, or rapid departures from the area) by listed whales, Steller sea lions or leatherback sea turtles are observed outside of the 160 dB, 180 dB, or 166 dB re 1 μPa isopleths, respectively, while airguns are operating, incidental take may be exceeded. If such reactions by listed species are observed while sonars are in operation, this may constitute take that is not covered in this Incidental Take Statement. The United States Geological Survey and the NMFS' Permits, Conservation, and Education Division must contact the Endangered Species Division to determine whether reinitation of consultation is required because of such operations.

Any incidental take of fin whales, humpback whales, sperm whales, Steller sea lions or leatherback sea turtles is restricted to the permitted action as proposed. If the actual incidental take meets or exceeds the predicted level, the United States Geological Survey and NMFS' Permits, Conservation and Education Division must reinitiate consultation. All anticipated takes would be "takes by harassment", as described previously, involving temporary changes in behavior.

Reasonable and Prudent Measures

NMFS believes the reasonable and prudent measures described below are necessary and appropriate to minimize the amount of incidental take of listed whales, Steller sea lions and leatherback sea turtles resulting from the proposed action. These measures are non-discretionary and must be binding conditions of the United States Geological Survey and NMFS' authorization

for the exemption in section 7(o)(2) to apply. If the United States Geological Survey or NMFS fail to ensure compliance with these terms and conditions, the protective coverage of section 7(o)(2) may lapse.

1. The United States Geological Survey must implement and monitor the effectiveness of mitigation measures incorporated as part of the proposed authorization of the incidental taking of fin, humpback and sperm whales and Steller sea lions pursuant to section 101(a)(5)(D) of the MMPA. Included are monitoring and reporting requirements for leatherback sea turtles as specified below.

Terms and Conditions

In order to be exempt from the prohibitions of section 9 of the ESA, the United States Geological Survey; NMFS' Permits, Conservation and Education Division; and L-DEO must comply with the following terms and conditions, which implement the Reasonable and Prudent Measure described above. These terms and conditions are non-discretionary.

To implement the Reasonable and Prudent Measures, the United States Geological Survey and NMFS shall ensure that:

1. <u>Monitoring Requirements</u>

(a) USGS shall utilize two, NMFS-qualified, vessel-based Protected Species Visual Observers (PSVOs) (except during meal times and restroom breaks, when at least one PSVO shall be on watch) are required to visually watch for and monitor marine mammals and leatherback sea turtles near the seismic source vessel during daytime airgun operations (from nautical twilight-dawn to nautical twilight-dusk) and before and during start-ups of airguns day or night. The *Langseth*'s vessel crew shall also assist in detecting marine mammals, when practicable. PSVOs shall have access to reticle binoculars (7x50 Fujinon), big-eye binoculars (25x150), and night vision devices. PSVO shifts shall last no longer than 4 hours at a time. PSVOs shall also make observations during daytime periods when the seismic system is not operating for comparison of animal abundance and behavior, when feasible.

(b) PSVOs shall conduct monitoring while the airgun array and streamer(s) are being deployed or recovered from the water.

(c) PSVOs shall record the following information when a marine mammal or leatherback sea turtle is sighted:

(i) species, group size, age/size/sex categories (if determinable), behavior when first sighted and after initial sighting, heading (if consistent), bearing and distance from seismic vessel, sighting cue, apparent reaction to the airguns or vessel (e.g., none, avoidance, approach, paralleling, etc., and including responses to ramp-up), and behavioral pace; and

(ii) time, location, heading, speed, activity of the vessel (including number of airguns operating and whether in state of ramp-up or power-down), Beaufort sea state and wind force, visibility, and sun glare; and

(iii) the data listed under Condition 1(c)(ii) shall also be recorded at the start and end of each observation watch and during a watch whenever there is a change in one or more of the variables.

(d) USGS shall utilize the passive acoustic monitoring (PAM) system, to the maximum extent practicable, to detect and allow some localization of marine mammals around the *Langseth* during all airgun operations and during most periods when airguns are not operating. One NMFS-qualified Protected Species Observer (PSO) and/or expert bioacoustician (i.e., Protected Species Acoustic Observer [PSAO]) shall monitor the PAM at all times in shifts no longer than 6 hours. An expert bioacoustician shall design and set up the PAM system and be present to operate or oversee PAM, and available when technical issues occur during the survey.

(e) PSAOs shall record the following when an animal is detected by the PAM:

(i) notify the on-duty PSVO(s) immediately of a vocalizing marine mammal so a power-down or shut-down can be initiated, if required;

(ii) enter the information regarding the vocalization into a database. The data to be entered include an acoustic encounter identification number, whether it was linked with a visual sighting, date, time when first and last heard and whenever any additional information was recorded, position, and water depth when first detected, bearing if determinable, species or species group (e.g., unidentified dolphin, sperm whale), types and nature of sounds heard (e.g., clicks, continuous, sporadic, whistles, creaks, burst pulses, strength of signal, etc.), and any other notable information.

(f) PSVOs shall visually observe the entire extent of the exclusion zone (EZ) (180 dB re 1 μPa [rms] for cetaceans and leatherback sea turtles and 190 dB re 1 μPa [rms] for pinnipeds using NMFS-qualified PSVOs, for at least 30 minutes (min) prior to starting the airgun array (day or night). If the PSVO finds a marine mammal or leatherback sea turtle within the EZ, USGS must delay the seismic survey until the animal(s) has left the area. If the PSVO sees a marine mammal that surfaces, then dives below the surface, the PSVO shall wait 30 min. If the PSVO sees a leatherback sea turtle that surfaces, then dives below the surface, the PSVO shall wait 8 min. If the PSVO sees no marine mammals or leatherback sea turtle during the applicable time, they should assume that the animal has moved beyond the EZ. If for any reason the entire radius cannot be seen for the entire 30 min (i.e., rough seas, fog, darkness), or if marine mammals are near, approaching, or in the EZ, the airguns may not be ramped-up. If one airgun is already running at a source level of at least 180 dB re 1 μPa (rms), USGS may start the second airgun without observing the entire EZ for 30 min prior, provided no marine mammals or

leatherback sea turtles are known to be near the EZ (in accordance with Condition 1[h] below).

(g) USGS shall establish a 180 dB re 1 μPa (rms) and 190 dB re 1 μPa (rms) EZ for marine mammals and sea turtles before the 4-string airgun array (6,600 in^3) is in operation; and a 180 dB re 1 μPa (rms) and 190 dB re 1 μPa (rms) EZ for Steller sea lions before a single airgun (40 in^3) is in operation, respectively.

(h) Implement a "ramp-up" procedure when starting up at the beginning of seismic operations or anytime after the entire array has been shutdown for more than 8 min, which means start the smallest gun first and add airguns in a sequence such that the source level of the array shall increase in steps not exceeding approximately 6 dB per 5-min period. During ramp-up, the PSVOs shall monitor the EZ, and if marine mammals or leatherback sea turtles are sighted, a power-down, or shut-down shall be implemented as though the full array were operational. Therefore, initiation of ramp-up procedures from shut-down requires that the PSVOs be able to view the full EZ as described in Condition 7(f) (above).

(i) Alter speed or course during seismic operations if a marine mammal or leatherback sea turtle, based on its position and relative motion, appears likely to enter the relevant EZ. If speed or course alteration is not safe or practicable, or if after alteration the marine mammal or sea turtle still appears likely to enter the EZ, further mitigation measures, such as a power-down or shut-down, shall be taken.

(j) Power-down or shut-down the airgun(s) if a marine mammal or leatherback sea turtle is detected within, approaches, or enters the relevant EZ. A shut-down means all operating airguns are shut-down (i.e., turned off). A power-down means reducing the number of operating airguns to a single operating 40 in^3 airgun, which reduces the EZ to the degree that the animal(s) is no longer in or about to enter it.

(k) Following a power-down, if the marine mammal or Leatherback sea turtle approaches the smaller designated EZ, the airguns must then be completely shut-down. Airgun activity shall not resume until the PSVO has visually observed the marine mammal(s) exiting the EZ and is not likely to return, or has not been seen within the EZ 30 min. For sea turtles airgun activity shall not resume until the PSVO has visually observed the animal(s) exiting the EZ and is not likely to return, or has not been seen within the EZ 8 min.

(l) Following a power-down or shut-down and subsequent animal departure, airgun operations may resume following ramp-up procedures described in Condition 1(h).

(m) Marine geophysical surveys may continue into night and low-light hours if such segment(s) of the survey is initiated when the entire relevant EZs are visible and can be effectively monitored.

(n) No initiation of airgun array operations is permitted from a shut-down position at night or during low-light hours (such as in dense fog or heavy rain) when the entire relevant EZ cannot be effectively monitored by the PSVO(s) on duty.

(o) If a North Pacific right (*Eubalaena japonica*), sei (*Balaenoptera borealis*) or , blue (*Balaenoptera musculus*), whale is visually sighted, the airgun array shall be shut-down regardless of the distance of the animal(s) to the sound source. The array shall not resume firing until 30 min after the last documented whale visual sighting.

(p) Concentrations of humpback (*Megaptera novaeangliae*), fin (*Balaenoptera physalus*), and/or killer (*Orcinus orca*) whales shall be avoided, if possible, and the array shall be powered-down if necessary. A concentration or group of whales shall consist of when three or more individuals are visually sighted and do not appear to be traveling (e.g., feeding, socializing, etc.).

(q) To the maximum extent practicable, schedule seismic operations (i.e., shooting airguns) during daylight hours and OBS operations (i.e., deploy/retrieve) to nighttime hours.

2. Reporting Requirements

USGS is required to:

(a) Submit a draft report on all activities and monitoring results to the Office of Protected Resources, NMFS, within 90 days of the completion of the survey. This report must contain and summarize the following information:

(i) Dates, times, locations, heading, speed, weather, sea conditions (including Beaufort sea state and wind force), and associated activities during all seismic operations and marine mammal sightings;

(ii) Species, number, location, distance from the vessel, and behavior of any marine mammals, as well as associated seismic activity (number of power-downs and shut-downs), observed throughout all monitoring activities.

(iii) An estimate of the number (by species) of marine mammals and leatherback sea turtles that: (A) are known to have been exposed to the seismic activity (based on visual observation) at received levels greater than or equal to 160 dB re 1 μPa (rms) and/or 180 dB re 1 μPa (rms) for cetaceans and sea turtles and 190 dB re 1 μPa (rms) for pinnipeds with a discussion of any specific behaviors those individuals exhibited; and (B) may have been exposed (based on reported and corrected empirical values for the 36 airgun array and modeling measurements for the single airgun) to the seismic activity at received levels greater than or equal to 160 dB re 1 μPa (rms) and/or 180 dB re 1 μPa (rms) for cetaceans and leatherback sea turtles and 190 dB re 1 μPa (rms) for pinnipeds with a discussion of the nature

of the probable consequences of that exposure on the individuals that have been exposed.

(iv) A description of the implementation and effectiveness of the: (A) terms and conditions of this Incidental Take Statement and mitigation measures of the Incidental Harassment Authorization. This report shall confirm the implementation of each Term and Condition, as well as any conservation recommendations, and describe their effectiveness, for minimizing the adverse effects of the action on ESA-listed marine mammals.

3. In the unanticipated event that the specified activity clearly causes the take of a marine mammal in a manner such as an injury (Level A harassment), serious injury or mortality (e.g., ship-strike, gear interaction, and/or entanglement), the United States Geological Survey shall immediately cease the specified activities and immediately report the incident to the Chief of the Permits, Conservation, and Education Division, Office of Protected Resources, NMFS and the Alaska Regional Stranding Coordinators. The report must include the following information:

(a) time, date, and location (latitude/longitude) of the incident; the name and type of vessel involved; the vessel's speed during and leading up to the incident; description of the incident; status of all sound source use in the 24 hours preceding the incident; water depth; environmental conditions (e.g., wind speed and direction, Beaufort sea state, cloud cover, and visibility); description of marine mammal observations in the 24 hours preceding the incident; species identification or description of the animal(s) involved; the fate of the animal(s); and photographs or video footage of the animal (if equipment is available).

Activities shall not resume until NMFS is able to review the circumstances of the prohibited take. NMFS shall work with USGS to determine what is necessary to minimize the likelihood of further prohibited take and ensure MMPA compliance. USGS may not resume their activities until notified by NMFS via letter or email, or telephone.

In the unanticipated event that the specified activity clearly causes the take of a leatherback sea turtle by injury or mortality (e.g., ship-strike, gear interaction, and/or entanglement), the United States Geological Survey shall immediately report the incident to the Chief of the Permits, Conservation, and Education Division, Office of Protected Resources, the Chief of the Endangered Species Division, Office of Protected Resources and the Alaska Regional Stranding Coordinators. The report must include the same information contained in 3(a) above.

In the event that USGS discovers an injured or dead marine mammal or leatherback sea turtle and the lead PSO determines that the cause of the injury or death is unknown and the death is relatively recent (i.e., in less than a moderate state of decomposition as described in the next paragraph), USGS will immediately report the incident to the Chief of the Permits Conservation, and Education Division, Office of Protected Resources, NMFS, at 301-713-2289, and the NMFS Alaska Stranding Hotline (1-877-925-7773) and/or by email to the Alaska Regional Stranding Coordinators. The report must include the same information identified in the paragraph 3(a)

above. Activities may continue while NMFS reviews the circumstances of the incident. NMFS will work with USGS to determine whether modifications in the activities are appropriate.

In the event that USGS discovers an injured or dead marine mammal or leatherback sea turtle, and the lead PSO determines that the injury or death is not associated with or related to the survey activities, USGS shall report the incident to the Chief of the Permits, Conservation, and Education Division, Office of Protected Resources, NMFS, at 301-713-2289, and the NMFS Alaska Stranding Hotline (1-877-925-7773) and/or by email to the Alaska Regional Stranding Coordinators within 24 hours of the discovery. USGS shall provide photographs or video footage (if available) or other documentation of the stranded animal sighting to NMFS and the Marine Mammal or Sea Turtle Stranding Network.

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 endangered and threatened species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on listed species or critical habitat, to help implement recovery plans, or to develop information.

We recommend the following conservation recommendations, which would provide information for future consultations involving seismic surveys and the issuance of incidental harassment authorizations that may affect endangered large whales and endangered or threatened sea turtles:

1. *Improve estimates of levels and forms of "take" and responses to seismic sounds.* The Permits, Conservation and Education Division should review reports submitted for this and other prior geophysical research surveys funded by the National Science Foundation and conducted by the United States Geological Survey and compile and analyze information to improve agency estimates of the number of the different species of marine mammals and sea turtles that are likely to be exposed to sounds from seismic surveys, the response of those species to this exposure, and the probable consequences of those responses on the life history of individual animals. The results should be provided to the Endangered Species Division as part of requests for consultation on future proposals to authorize incidental harassment.

In order for the NMFS' Office of Protected Resources Endangered Species Division to be kept informed of actions minimizing or avoiding adverse effects on, or benefiting, ESA-listed species or their habitats, the NMFS' Office of Protected Resources Permits, Conservation and Education Division should notify the Endangered Species Division of any conservation recommendations they implement in their final action.

REINITIATION NOTICE

This concludes formal consultation on the proposed marine geophysical survey conducted by the United States Geological Survey on board the *R/V Langseth* in the Gulf of Alaska, and the issuance of an incidental harassment authorization for the proposed survey pursuant to section 101(a)(5)(D) of the Marine Mammal Protection Act (MMPA). As provided in 50 CFR §402.16, 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 gency action that may affect listed species or critical habitat in a manner or to an extent not considered in this opinion; (3) the agency action is subsequently modified in a manner that causes an effect to the listed species or critical habitat not considered in this opinion; or (4) a new species is listed or critical habitat designated that may be affected by the action. In instances where the amount or extent of authorized take is exceeded, section 7 consultation must be reinitiated immediately.

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