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

Biological Opinion

Agencies: National Science Foundation-Division of Ocean Sciences and NOAA's National Marine Fisheries Service-Office of Protected Resources-Permits and Conservation Division

Activities Considered: Seismic survey by the Lamont-Doherty Earth Observatory in the northwestern Pacific Ocean over the Shatsky Rise and Issuance of an Incidental Harassment Authorization pursuant to Section 101(a)(5)(D) of the Marine Mammal Protection Act (MMPA)

Consultation Conducted by: NOAA's National Marine Fisheries Service-Office of Protected Resources-Endangered Species Act Interagency Cooperation Division

Approved by: ______________________________

Date: 3/23/12

Section 7(a)(2) of the Endangered Species Act (ESA)(16 U.S.C. 1531 *et seq.*) requires that each federal agency shall insure 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 designated for them, that agency is required to consult with either the NOAA's National Marine Fisheries Service (NMFS) or the U.S. Fish and Wildlife Service, depending upon the listed resources that may be affected. For the actions described in this document, the action agencies are the National Science Foundation (NSF), which proposes to fund the Lamont-Doherty Earth Observatory (L-DEO) to conduct a seismic survey in the northwest Pacific Ocean over the Shatsky Rise from March to May of 2012 and the NMFS' Office of Protected Resources-Permits and Conservation Division, which proposes to authorize the NSF and L-DEO to "take" marine mammals incidental to those seismic surveys. The consulting agency for these proposals in the NMFS' Office of Protected Resources – Endangered Species Act Interagency Cooperation Division.

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(IHA) application, draft IHA, environmental assessment, monitoring reports from similar activities, published and unpublished scientific information on endangered and threatened species and their surrogates, scientific and commercial information such as reports from government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

Consultation history

On November 1, 2011, the NMFS' Permits and Conservation Division received an application for L-DEO to incidentally harass marine mammal and sea turtle species during a seismic survey cruise over the Shatsky Rise. On the same date, the NMFS' Endangered Species Act Interagency Cooperation Division received a request for formal consultation from the NSF to incidentally harass marine mammal and sea turtle species during a seismic survey cruise over the Shatsky Rise which was sufficient to initiate consultation with the NSF.

On January 24, 2012, the Endangered Species Act Interagency Cooperation Division received a request for formal consultation from the Permits and Conservation Division to authorize incidental harassment of marine mammals during a seismic survey cruise over the Shatsky Rise. Information was sufficient to initiate consultation with the Permits and Conservation Division on this day.

Description of the proposed actions

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.

The NSF proposes to fund the L-DEO to conduct a seismic survey in the northwestern Pacific Ocean over the Shatsky Rise from roughly 24 March-7 May, 2012. It is possible that temporary delays could occur due to weather, equipment malfunctions, or other unforeseen circumstances. Thus, the IHA is proposed to be effective until May 7, 2012. The *R/V Marcus G. Langseth* (*Langseth*) would conduct the survey. The *Langseth* would deploy an array of 36 airguns as an energy source. In addition, a multibeam echosounder and a sub-bottom profiler would continuously operate from the *Langseth*. The *Langseth* would also deploy a 6 km-long hydrophone streamer. The Permits and Conservation Division proposes to issue an IHA for takes of marine mammals that would occur incidental to these studies, pursuant to Section 101(a)(5)(D) of the Marine Mammal Protection Act (MMPA), 16 U.S.C. §1371 (a)(5)(D).

The purpose of the proposed activities is to understand the crustal structure of the Shatsky Rise and to address questions of planetary history and magnetism, tectonics, and earthquake occurrence and distribution.

The survey would occur exclusively in the high seas outside of any territorial claims. All planned geophysical data acquisition activities would be conducted by L-DEO with on-board assistance of the scientists who have proposed the study.

The planned seismic survey would consist of ~1,216 km of survey lines, all in water >1,000 m deep.

Schedule

The *Langseth* is scheduled to depart Yokohama, Japan on or about 24 March 2012 for the study area (Figure 1). Seismic surveys should take about seven days, ending on roughly 9 April 2012. Following completion of the survey, the *Langseth* would return to Honolulu on or about 16 April 2012.

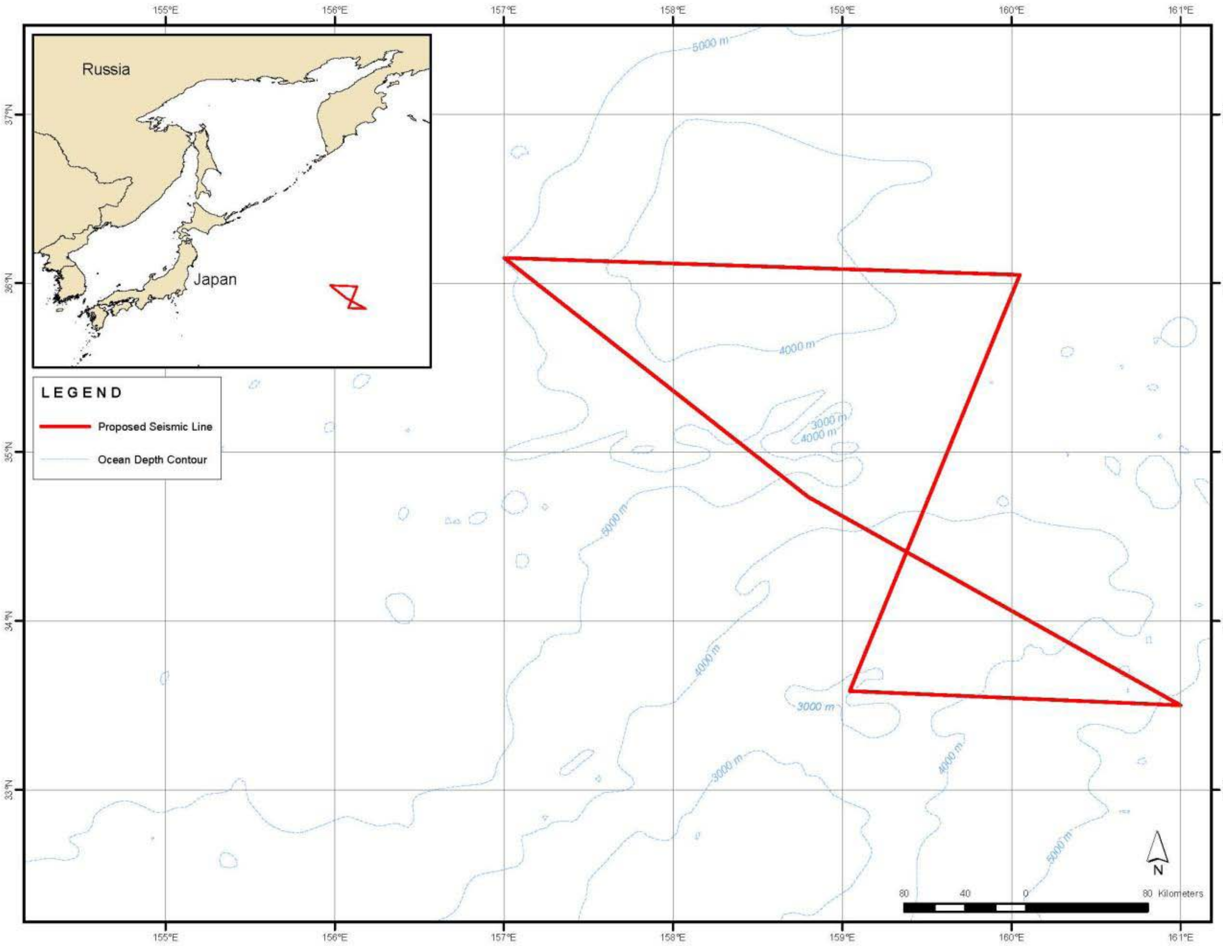

Figure 1. Study area and proposed seismic transect over the Shatsky Rise for the L-DEO survey planned for March-May 2012.

Source vessel specifications

The *Langseth* would tow the 36-airgun array along predetermined lines (Figure 1). The *Langseth's* design is that of a seismic research vessel, with a particularly quiet propulsion system to avoid interference with the seismic signals. The operating speed during seismic acquisition is typically 7.4–9.3 km/h. When not towing seismic survey gear, the *Langseth* can cruise at 20–24 km/h. The *Langseth* would also serve as the platform from which marine mammal and sea turtle observers (MMOs) would watch for animals.

Airgun description

The airgun array would consist of 36 airguns, with a total volume of ~6,600 in^3, including Bolt 1500LL and Bolt 1900LLX airguns. The airgun configuration includes four identical linear arrays or "strings" (Figure 2). Each string would have ten airguns. Nine airguns in each string would fire simultaneously, with the tenth kept in reserve as a spare. The four airgun strings would be towed ~100 m behind the vessel. The tow depth of the array would be 9 m. The airgun array would fire every 20 s while conducting multichannel seismic surveys. During firing, a brief (~0.1 s) pulse of sound would be emitted, but be silent during the intervening periods.

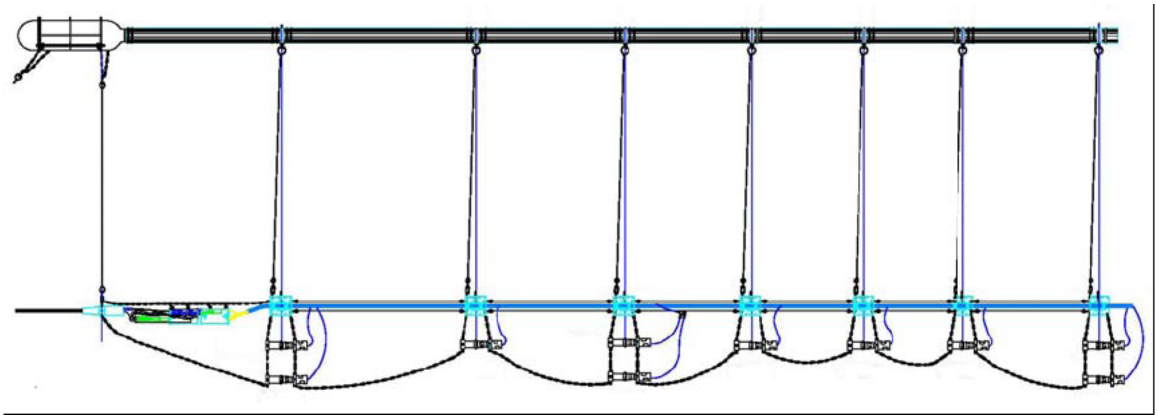

Figure 2. One linear airgun array or string with ten airguns, nine of which would be operating.

36-airgun array specifications

- Energy source	36-1,900 psi bolt airguns of 40–360 in^3 each, in four strings of nine operating airguns per string
- Source output (downward)	0-pk is 84 bar-m (259 dB re 1 μPa·m); pk-pk is 177 bar·m (265 dB)
- Air discharge volume	~6,600 in^3
- Dominant frequency components	2–188 Hz

Because the actual source originates from 36 airguns rather than a single point source, the highest sound levels measurable at any location in the water is less than the nominal source level. In addition, the effective source level for sound propagating in near-horizontal directions would be substantially lower than the nominal source level applicable to downward propagation because of the directional nature of the sound from the airgun array.

Multibeam echosounder and sub-bottom profiler

Along with airgun operations, two additional acoustical data acquisition systems would operate during the survey. The multibeam echosounder and sub-bottom profiler systems would map the ocean floor during the survey. These sound sources would operate from the *Langseth* simultaneously with the airgun array.

The multibeam echosounder is a hull-mounted system operating at 10.5-13 kHz. The beamwidth is 1 or 2° fore–aft and 150° perpendicular to the ship's line of travel. The maximum source level is 242 dB re 1 $\mu Pa \cdot m_{rms}$. For deepwater operation, each "ping" consists of eight successive fan-shaped transmissions, each 2 to 15 ms in duration and each ensonifying a sector that extends 1° fore–aft. The eight successive transmissions span an overall cross-track angular extent of about 150°, with 2 ms gaps between the pulses for successive sectors (Maritime 2005).

The sub-bottom profiler provides information about the sedimentary features and the bottom topography that is being mapped simultaneously by the multibeam echosounder. The output varies with water depth from 50 watts in shallow water to 1,000 (204 dB) watts in deep water. The pulse interval is 1 s, but a common mode of operation is to broadcast five pulses at 1-s intervals followed by a 5-s pause.

Langseth sub-bottom profiler specifications

Maximum/normal source output (downward)	204 dB re 1 μPa·m; 800 watts
Dominant frequency component	3.5 kHz
Bandwidth	1.0 kHz with pulse duration 4 ms 0.5 kHz with pulse duration 2 ms 0.25 kHz with pulse duration 1 ms
Nominal beam width	30º
Pulse duration	1, 2, or 4 ms

Proposed exclusion zones

Predicted sound levels vs. distance and depth. The L-DEO has predicted received sound levels, in relation to distance and direction from a single 1900LL 40-in^3 airgun used during power-downs (Figure 3). Empirical data concerning 180, 170, and 160 dB re 1 μPa$_{rms}$ distances were acquired during the acoustic calibration study of the *Langseth*'s 36-airgun 6,600 in^3 array in 2007-2008 (Tolstoy et al. 2009). Results of the propagation measurements (Tolstoy et al. 2009) showed that radii around the airguns for various received levels varied with water depth. However, the depth of the array was different in the Gulf of Mexico calibration study (6 m) than in the proposed survey (9 m). Because propagation varies with array depth, correction factors have been applied to the distances reported by Tolstoy et al. (2009). The correction factors used were the ratios of the 160-, 170-, 180-, and 190-dB distances from the modeled results for the 6600-in^3 airgun array towed at 6-m and 9-m depths.

Table 1 shows the distances at which four rms (root mean squared) sound levels are expected to be received from the 36-airgun array and a single airgun. The 180 and 190 dB re 1 μPa$_{rms}$ distances are the safety criteria as specified by NMFS (1995) and are applicable to cetaceans and pinnipeds, respectively. The 180 dB distance would also be used as the exclusion zone (EZ) for sea turtles, as required by the NMFS during most other recent L-DEO seismic projects (Holst and Beland 2008; Holst and Smultea 2008b; Holst et al. 2005a; Holt 2008; Smultea et al. 2004).

Table 1. Predicted distances to which sound levels ≥190, 180, 170, and 160 dB re 1 μPa$_{rms}$ could be received in deep water (>1,000 m) from the 36-airgun array, as well as a single airgun.

Source and volume	**Tow depth (m)**	**Predicted rms radii (m)**			
		190 dB	**180 dB**	**170 dB**	**160 dB**
Single bolt airgun 40 in^3	9	12	40	120	385
4 strings 36 airguns 6,600 in^3	9	400	940	2200	3850

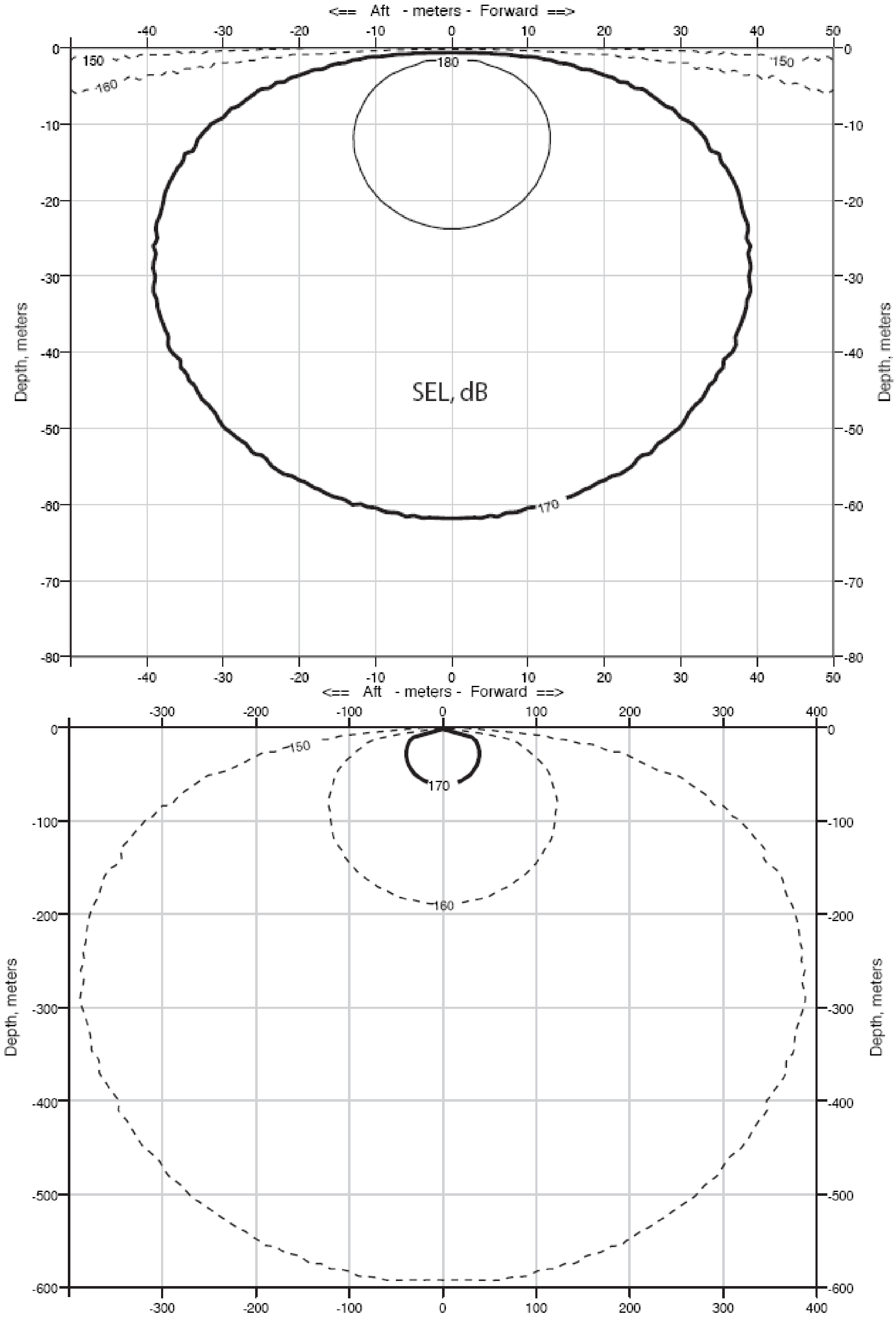

Figure 3. Modeled received sound levels (SELs) from a single 40-in^3 airgun operating in deep water at a 9-m tow depth. Received rms levels (SPLs) are likely ~10 dB higher.

Incidental Harassment Authorization

The NMFS' Permits and Conservation Division is proposing to issue an IHA authorizing non-lethal "takes" by harassment of marine mammals incidental to the planned seismic survey, pursuant to Section 101 (a)(5)(D) of the Marine Mammal Protection Act (MMPA), 16 U.S.C. § 1371 (a)(5)(D). The IHA would be valid from 24 March through 7 May 2012, and would authorize the incidental harassment of the following endangered species (among other species): blue whales (*Balaenoptera musculus*), fin whales (*Balaenoptera physalus*), humpback whales (*Megaptera novaeangliae*), North Pacific right whales (*Eubalaena japonica*), sei whales (*Balaenoptera borealis*), sperm whales (*Physeter macrocephalus*), and other non-listed marine mammals. The proposed IHA identifies the following requirements that L-DEO must comply with as part of its authorization.

A. Establish a safety radius corresponding to the anticipated 180-dB isopleth for full (6,600 in^3) and single (40 in^3) airgun operations.

B. Use two, NMFS-approved, vessel-based MMOs to watch for and monitor marine mammals near the seismic source vessel during daytime airgun operations, start-ups of airguns at night, and while the seismic array is being deployed and retrieved. Vessel crew will also assist in detecting marine mammals, when practical. Observers will have access to reticle binoculars (7 X 50 Fujinon), big-eye binoculars (25 X 150), and night vision devices (ITT F500 Series Generation 3 binocular-image intensifier or equivalent). Laser range finding binoculars (Leica LRF 1200 laser rangefinder or equivalent) will be available to assist with distance estimation. MMOs shifts will last no longer than 4 hours at a time. MMOs will also observe during daytime periods when the seismic system is not operating for comparisons of animal abundance and behavior, when feasible.

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.
>
> ii. Time, location, heading, speed, activity of the vessel (including number of airguns operating and whether in state of ramp-up or power-down), sea state, visibility, cloud cover, and sun glare.
>
> iii. The data listed under ii. would 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. Visually observe the entire extent of the safety radius using MMOs, for at least 30 min prior to starting the airgun (day or night). If the MMO finds a marine mammal within the safety zone, L-DEO must delay the seismic survey until the marine mammal has left the area. If the MMO sees a marine mammal that surfaces, then dives below the surface, the observer shall wait 30 minutes. If the MMO sees no marine mammals during that time, they should assume that the animal has moved beyond the safety zone. 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 safety radius, the airguns may not be started up. If one airgun is already running at a source level of at least 180 dB, L-DEO may start subsequent guns without observing the entire safety radius for 30 min prior, provided no marine mammals are known to be near the safety radius.

E. Use the passive acoustic monitoring system (PAM) to detect marine mammals around the *Langseth* during all airgun operations and during most periods when airguns are not operating. One MMO and/or bioacoustician will monitor the PAM at all times in shifts of 1-6 h. A 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.

F. Do and record the following information when an animal is detected by the PAM:

i. contact the MMO immediately (and initiate power or shut-down, 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, types and nature of sounds heard (e.g., clicks, continuous, sporadic, whistles, creaks, burst pulses, strength of signal, etc.), and any other notable information.

G. Apply a “ramp-up” procedure when starting up at the beginning of seismic operations or anytime after the entire array has been shut down 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 will increase in steps not exceeding approximately 6 dB per 5-min period (total duration of 35 min to full power). During ramp-up, the MMOs will monitor the safety radius, and if marine mammals are sighted, a course/speed alteration, power-down, or shut-down will occur as though the full array were operational.

H. Alter speed or course during seismic operations if a marine mammal, based on its position and relative motion, appears likely to enter the safety zone. If speed or course alteration is not safe or practical, or if after alteration the marine mammal still appears likely to enter the safety zone, further mitigation measures, such as power-down or shut-down, will be taken.

I. Shut-down or power-down the airguns upon marine mammal detection within, approaching, or entering the safety radius. A power-down means shutting down one or more airguns and reducing the safety radius to the degree that the animal is outside of it. Following a power-down, if the marine mammal approaches the smaller designated safety radius, the airguns must completely shut down. Airgun activity will not resume until the marine mammal has cleared the safety radius, which means it was visually observed to have left the safety radius, or has not been seen within the radius for 15 min (small odontocetes) or 30 min (mysticetes and large odontocetes). If a North Pacific right whale is sighted, airguns will be shut down immediately. The array will not resume firing until 30 min after the last documented whale visual sighting.

J. Emergency shutdown. In the unanticipated event that any taking of a marine mammal in a manner prohibited by the proposed Authorization occurs, such as an injury, serious injury or mortality, and is judged to result from these activities, L-DEO will immediately cease operating all authorized sound sources and report the incident to the Chief of the Permits and Conservation Division, Office of Protected Resources, NMFS, at 301-427-8401. L-DEO will postpone the research activities until NMFS is able to review the circumstances of the take. NMFS will work with L-DEO to determine whether modifications in the activities are appropriate and necessary, and notify L-DEO that they may resume the seismic survey operations.

K. In the unanticipated event that any cases of marine mammal injury or mortality are judged to result from these activities, L-DEO will cease operating seismic airguns and report the incident to the Office of Protected Resources, NMFS, immediately. Airgun operation will then be postponed until NMFS is able to review the circumstances and work with L-DEO to determine whether modifications in the activities are appropriate and necessary.

L. Conduct seismic operations during daylight hours where possible.

M. L-DEO is required to comply with the Terms and Conditions of the Opinion's Incidental Take Statement issued to both the NSF and the NMFS' Office of Protected Resources.

In addition, the proposed IHA requires L-DEO to adhere to the following reporting requirements:

A. The Holder of this Authorization is required to submit a report on all activities and monitoring results to the Office of Protected Resources, NMFS, within 90 days after the expiration of the IHA. This report must contain and summarize the following information:

> i. Dates, times, locations, heading, speed, weather, and associated activities during all seismic operations;
>
> 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 shutdowns), 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 (visual observation) at received levels greater than or equal to 160 dB re 1 microPa (rms) and/or 180 dB re 1 microPa (rms) with a discussion of any specific behaviors those individuals exhibited and
>
> b. may have been exposed (modeling results) to the seismic activity at received levels greater than or equal to 160 dB re 1 microPa (rms) and/or 180 dB re 1 microPa (rms) 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 Opinion's Incidental Take Statement, and
>
> b. mitigation measures of the IHA. For the Opinion, the report will confirm the implementation of each term and condition and describe the effectiveness, as well as any conservation measures, for minimizing the adverse effects of the action on listed whales.

Approach to the assessment

The 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., Anderson 2000; Brandon 1978; Mills and Beatty 1979; Stearns 1992). 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 listed resources* 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 consists of monitoring reports submitted by past and present permit holders, reports from NMFS Science Centers; reports prepared by natural resource agencies in States and other countries, reports from non-governmental organizations involved in marine conservation issues, the information provided by the Permits and Conservation 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 former 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, Ingenta Connect, Aquatic Sciences and Fisheries Abstracts, JSTOR, Conference Papers Index, First Search (Article First, ECO, WorldCat), Web of Science, Oceanic Abstracts, Google Scholar, and Science Direct.

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 will exhibit a particular response to acoustic exposure or close vessel approach) as well as data that do not support that conclusion. When data were equivocal or when faced with substantial uncertainty, our decisions are 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).

In this particular assessment, we identified the stressors associated with the action and evaluated which had a significant possibility of occurring based upon previous seismic surveys. Of the probable stressors, we identified the species that were expected to co-occur with the effects of the action, particularly the acoustic isopleths of the airgun and other sound sources. Utilizing survey data from previous years, density estimates per unit area of listed whales were multiplied by the area to be ensonified where effects were expected.

In the process of this assessment, we were required to make several assumptions where data were insufficient to support conclusions regarding the specific species and actions at hand. These included:

- Baleen whales can generally hear low-frequency sound better than high frequencies, as the former is the primarily the range in which they vocalize. Humpback whales frequently vocalize with mid-frequency sound and are likely to hear at these frequencies as well. Because of this, we can partition baleen whales into two groups: those that are specialists at hearing low frequencies (ex.: blue, fin, right, and sei whales) and those that hear at low- to mid-frequencies (humpback whales). Toothed whales (such as sperm whales) are better adapted to hear mid- and high-frequency sound for the same reason (although this species also responds to low-frequency sound and is considered to hear at low-, mid-, and high frequencies). Sperm whales are also assumed to have similar hearing qualities as other, better studied, toothed whales. Hearing in sea turtles is generally similar within the taxa, with data from loggerhead and green sea turtles being representative of the taxa as a whole.
- Species for which little or no information on response to sound will respond similarly to their close taxonomic or ecological relatives (i.e., baleen whales respond similarly to each other; same for sea turtles).

Action area

The proposed seismic survey should occur roughly 1,200 km east of Japan between 24 March and 7 May, 2012 (Figure 1 on page 3). The survey would encompass deep water in an area from 33–36° N and 156–161° E in the west-central Pacific Ocean. Responses to seismic sound sources by listed species occur within the 160 dB isopleths (modeled to be 3.85 km from the *Langseth*). This expands the action area beyond the seismic survey track lines (1,216 km) to an ensonified region of 9,229 km^2, or 11,536 km^2 to account for additional effort resulting from delays, equipment failure, re-surveying, and other factors.

Status of listed resources

The NMFS has determined that the actions considered in this Opinion may affect species listed in Table 2, which are provided protection under the Endangered Species Act of 1973, as amended (16 U.S.C. 1531 *et seq*.).

Critical habitat has not been established in the region of the proposed action area. We conclude that critical habitat will not be impacted by the proposed actions.

The biology and ecology of species with anticipated exposure below informs the effects analysis for this Opinion. Summaries of the global status and trends of each species presented provide a foundation for the analysis of species as a whole.

Table 2. Listed species in the action area.

Common name	Scientific name	Status
Cetaceans		
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
Marine turtles		
Green sea turtle	*Chelonia mydas*	Endangered/ Threatened
Hawksbill sea turtle	*Eretmochelys imbricate*	Threatened
Leatherback sea turtle	*Dermochelys coriacea*	Endangered
Loggerhead sea turtle (North Pacific DPS)	*Caretta caretta*	Endangered
Olive ridley sea turtle	*Lepidochelys olivacea*	Endangered/ Threatened

Cetaceans

Blue whale

Description of the species. Blue whales occur primarily in the open ocean from tropical to polar waters worldwide. Blue whales are highly mobile, and their migratory patterns are not well known (Perry et al. 1999; Reeves et al. 2004). Blue whales migrate toward the warmer waters of the subtropics in fall to reduce energy costs, avoid ice entrapment, and reproduce (NMFS 1998a). Blue whales typically occur alone or in groups of up to five animals, although larger foraging aggregations of up to 50 have been reported including aggregations mixed with other rorquals such as fin whales (Corkeron et al. 1999; Shirihai 2002).

Subspecies. Several blue whale subspecies have been characterized from morphological and geographical variability, but the validity of blue whale subspecies designations remains uncertain (McDonald et al. 2006). The largest, the Antarctic or true blue whale (*Balaenoptera musculus intermedia*), occurs in the highest Southern Hemisphere latitudes (Gilpatrick and Perryman. 2009). During austral summers, “true” blue whales occur close to Antarctic ice. A slightly smaller blue whale, *B. musculus musculus*, inhabits the Northern Hemisphere (Gilpatrick and Perryman. 2009). The pygmy blue whale (B. *musculus brevicauda*), may be geographically distinct from *B. m. musculus* (Kato et al. 1995). Pygmy blue whales occur north of the Antarctic Convergence (60°-80° E and 66°-70° S), while true blue whales are south of the Convergence (58° S) in the austral summer (Kasamatsu et al. 1996; Kato et al. 1995). A fourth subspecies, *B. musculus indica,* may exist in the northern Indian Ocean (McDonald et al. 2006).

Population structure. Little is known about population and stock structure[1] of blue whales. Studies suggest a wide range of alternative population and stock scenarios based on movement, feeding, and acoustic data. Some suggest there are as many as 10 global populations, while others suggest that the species is composed of a single panmictic population (Gambell 1979; Gilpatrick and Perryman. 2009; Reeves et al. 1998). For management purposes, the International Whaling Commission (IWC) considers all Pacific blue whales to be a single stock, whereas under the MMPA, the NMFS recognizes four stocks of blue whales: western North Pacific Ocean, eastern North Pacific Ocean, Northern Indian Ocean, and Southern Hemisphere.

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

North Atlantic. Blue whales are found from the Arctic to at least mid-latitude waters, and typically inhabit the open ocean with occasional occurrences in the U.S. EEZ (Gagnon and Clark 1993; Wenzel et al. 1988; Yochem and Leatherwood 1985). Yochem and Leatherwood (1985) summarized records suggesting winter range extends south to Florida and the Gulf of Mexico. The U.S. Navy's Sound Surveillance System acoustic system has detected blue whales in much of the North Atlantic, including subtropical waters north of the West Indies and deep waters east of the U.S. Atlantic EEZ (Clark 1995). Blue whales are rare in the shelf waters of the eastern U.S. In the western North Atlantic, blue whales are most frequently sighted from the Gulf of St. Lawrence and eastern Nova Scotia and in waters off Newfoundland, during the winter (Sears et al. 1987). In the eastern North Atlantic, blue whales have been observed off the Azores, although Reiner et al. (1993) did not consider them common in that area. Observations of feeding have recently occurred over Ireland's western continental slope (Wall et al. 2009).

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

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[1] "Populations" herein are a group of individual organisms that live in a given area and share a common genetic heritage. While genetic exchange may occur with neighbouring populations, the rate of exchange is greater between individuals of the same population than among populations---a population is driven more by internal dynamics, birth and death processes, than by immigration or emigration of individuals. To differentiate populations, NMFS considers geographic distribution and spatial separation, life history, behavioral and morphological traits, as well as genetic differentiation, where it has been examined. In many cases, the behavioral and morphological differences may evolve and be detected before genetic variation occurs. In some cases, the term "stock" is synonymous with this definition of "population" while other usages of "stock" are not.

Leatherwood et al. 1982; Stewart et al. 1987), possibly supporting a return to historical migration patterns (Anonmyous. 2009).

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

Indian Ocean. Blue whale sightings have occurred in the Gulf of Aden, Persian Gulf, Arabian Sea, and across the Bay of Bengal to Burma and the Strait of Malacca (Clapham et al. 1999; Mikhalev 1997; Mizroch et al. 1984).

Southern Hemisphere. Blue whales range from the edge of the Antarctic pack ice (40^{o}-78^{o} S) during the austral summer north to Ecuador, Brazil, South Africa, Australia, and New Zealand during the austral winter (Shirihai 2002). Occurrence in Antarctic waters appears to be highest February-May as well as in November (Gedamke and Robinson. 2010; Sirovic et al. 2009). Gedamke and Robinson (2010) found blue whales to be particularly numerous and/or vocal north of Prydz Bay, Antarctica based upon sonobuoy deployments. Pygmy blue whales were also frequently heard in Antarctic waters, further south than they had previously been documented (Gedamke and Robinson 2010). Other than a single vocal record in Atlantic waters off Angola, pygmy blue whales have been exclusively documented in the Indian Ocean or western Pacific (Cerchio et al. 2010a; Mccauley and Jenner 2010).

Blue whales are occasionally sighted in pelagic waters off the western coast of Costa Rica and Nicaragua, near the Galápagos Islands, and along the coasts of Ecuador and northern Peru (Aguayo 1974; Clarke 1980b; Donovan 1984; LGL Ltd. 2007; Mate et al. 1999; Palacios 1999; Reilly and Thayer 1990). Individuals here may represent two populations; the true and pygmy blue whales of the Southern Hemisphere (Gilpatrick and Perryman. 2009), although, recent analyses of vocalizations and photos have linked blue whales found in the Costa Rica Dome to the North Pacific population (Chandler and Calambokidis 2004).

Age distribution. Blue whales may reach 70–80 years of age (COSEWIC 2002; Yochem and Leatherwood 1985).

Reproduction. Gestation takes 10-12 months, followed by a 6-7 month nursing period. Sexual maturity occurs at 5-15 years of age and calves are born at 2-3 year intervals (COSEWIC 2002; NMFS 1998b; Yochem and Leatherwood 1985). Recent data from illegal Russian whaling for Antarctic and pygmy blue whales support sexual maturity at 23 m and 19-20 m, respectively (Branch and Mikhalev 2008).

Movement. Satellite tagging indicates that, for blue whales tagged off Southern California, movement is more linear and faster (3.7 km/h) while traveling versus while foraging (1.7 km/h)(Bailey et al. 2009). Residency times in what are likely prey patches averages 21 days and constituted 29% of an individual's time overall, although foraging could apparently occur at any time of year for tagged individuals (Bailey et al. 2009). Broad scale movements also varied greatly, likely in response to oceanographic conditions influencing prey abundance and distribution (Bailey et al. 2009).

Feeding. Data indicate that some summer feeding takes place at low latitudes in upwelling-modified waters, and that some whales remain year-round at either low or high latitudes (Clarke and Charif 1998; Hucke-Gaete et al. 2004; Reilly and Thayer 1990; Yochem and Leatherwood 1985). One population feeds in California waters from June to November and migrates south in winter/spring (Calambokidis et al. 1990; Mate et al. 1999). Prey availability likely dictates blue whale distribution for most of the year (Burtenshaw et al. 2004; Clapham et al. 1999; Sears 2002 as cited in NMFS 2006a). The large size of blue whales requires higher energy requirements than smaller whales and potentially prohibits fasting Mate et al. (1999). Krill are the primary prey of blue whales in the North Pacific (Kawamura 1980; Yochem and Leatherwood 1985).

While feeding, blue whales show slowed and less obvious avoidance behavior then when not feeding (Sears et al. 1983 as cited in NMFS 2005b).

Diving. Blue whales spend greater than 94% of their time underwater (Lagerquist et al. 2000). Generally, blue whales dive 5-20 times at 12-20 sec intervals before a deep dive of 3-30 min (Croll et al. 1999; Leatherwood et al. 1976; Mackintosh 1965; Maser et al. 1981; Strong 1990; Yochem and Leatherwood 1985). Average foraging dives are 140 m deep and last for 7.8 min (Croll et al. 2001). Non-foraging dives are shallower and shorter, averaging 68 m and 4.9 min (Croll et al. 2001). However, dives of up to 300 m are known (Calambokidis et al. 2003). Nighttime dives are generally shallower (50 m).

Blue whales occur singly or in groups of two or three (Aguayo 1974; Mackintosh 1965; Nemoto 1964; Pike and MacAskie 1969; Ruud 1956; Slijper 1962). However, larger foraging aggregations, even with other species such as fin whales, are regularly reported (Fiedler et al. 1998; Schoenherr 1991)

Vocalization and hearing. Blue whales produce prolonged low-frequency vocalizations that include moans in the range from 12.5-400 Hz, with dominant frequencies from 16-25 Hz, and songs that span frequencies from 16-60 Hz that last up to 36 sec repeated every 1 to 2 min (see McDonald et al. 1995). Berchok et al. (2006) examined vocalizations of St. Lawrence blue whales and found mean peak frequencies ranging from 17.0-78.7 Hz. Reported source levels are 180-188 dB re 1μPa, but may reach 195 dB re 1μPa (Aburto et al. 1997; Clark and Ellison 2004; Ketten 1998; McDonald et al. 2001).

As with other baleen whale vocalizations, blue whale vocalization function is unknown, although numerous hypotheses exist (maintaining spacing between individuals, recognition, socialization, navigation, contextual information transmission, and location of prey resources; (Edds-Walton 1997; Payne and Webb 1971; Thompson et al. 1992). Intense bouts of long, patterned sounds are common from fall through spring in low latitudes, but these also occur less frequently while in summer high-latitude feeding areas. Short, rapid sequences of 30-90 Hz calls are associated with socialization and may be displays by males based upon call seasonality and structure.

Blue whale calls appear to vary between western and eastern North Pacific regions, suggesting possible structuring in populations (Rivers 1997; Stafford et al. 2001).

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

Status and trends. Blue whales (including all subspecies) were originally listed as endangered in 1970 (35 FR 18319), and this status continues since the inception of the ESA in 1973.

Table 3 contains historic and current estimates of blue whales by region. Globally, blue whale abundance has been estimated at between 5,000-13,000 animals (COSEWIC 2002; Yochem and Leatherwood 1985); a fraction of the 200,000 or more that are estimated to have populated the oceans prior to whaling (Maser et al. 1981; U.S. Department of Commerce 1983).

Table 3. Summary of past and present blue whale abundance.

Region	Population, stock, or study area	Pre-exploitation estimate	95% C.I.	Current estimate	95% C.I.	Source
Global	--	200,000	--	11,200-13,000	--	(DOC 1983; Maser et al. 1981)
				5,000-12,000		(COSEWIC 2002)
North Atlantic	Basinwide	1,100-1,500	--	100-555	--	(Braham 1991; Gambell 1976)
	NMFS - Western North Atlantic stock	--	--	308	--	(Sears et al. 1987)
North Pacific	Basinwide	4,900	--	1,400-1,900	--	(Gambell 1976)
				3,300	--	(Wade and Gerrodette 1993) and (Barlow 1997a) as combined in (Perry et al. 1999)
	Eastern Tropical Pacific	--	--	1,415	1,078-2,501	(Wade and Gerrodette 1993)
	EEZ of Costa Rica	--	--	48	22-102*	(Gerrodette and Palacios 1996)
	EEZs of Central America north of Costa Rica	--	--	94	34-257*	(Gerrodette and Palacios 1996)
	Eastern North Pacific	--	--	2,997	2,175-3,819*	(Calambokidis and Barlow 2004)
	NMFS - western North Pacific stock	--	--	n/a	--	(Carretta et al. 2006)
	NMFS - eastern North Pacific stock	--	--	2,842	CV=0.41	(Carretta et al. 2009)
Southern Hemisphere	Basinwide	150,000-210,000	--	5,000-6,000	--	(Gambell 1976; Yochem and Leatherwood 1985)
		300,000	--	--	--	(COSEWIC 2002)
		--	--	400-1,400	400-1,400	IWC, for years 1980-2000
		--	--	1,700	860-2,900	(IWC 2005c), point estimate for 1996
	Within IWC survey areas	--	--	1,255	--	(IWC 1996)
	Pygmy blue whale population	10,000	--	5,000	--	(Gambell 1976)
		13,000	--	6,500	--	(Zemsky and Sazhinov 1982)

*Note: Confidence Intervals (C.I.) not provided by the authors were calculated from Coefficients of Variation (C.V.) where available, using the computation from Gotelli and Ellison (2004).

North Atlantic. Commercial hunting had a severe effect on blue whales, such that they remain rare in some formerly important habitats, notably in the northern and northeastern North Atlantic (Sigurjónsson and Gunnlaugsson 1990). Sigurjónsson and Gunnlaugsson (1990) estimated that at least 11,000 blue whales were harvested from all whaling areas from the late nineteenth to mid-twentieth centuries. The actual size of the blue whale population in the North Atlantic is uncertain, but estimates range from a few hundred individuals to about 2,000 (Allen 1970; Mitchell 1974a; Sigurjónsson 1995; Sigurjónsson and Gunnlaugsson 1990). Current trends are unknown, although an increasing annual trend of 4.9% annually was reported for 1969–1988 off western and southwestern Iceland (Sigurjónsson and Gunnlaugsson 1990). Sigurjónsson and Gunnlaugsson (1990) concluded that the blue whale population had been increasing since the late 1950s. In the northeastern Atlantic, blue whales are most common west and south of Iceland and may be the largest concentration of blue whales in the North Atlantic (Pike et al. 2009b). In this area, the population may be recovering at a rate of 4-5% (Pike et al. 2009b). Punt (2010) estimated the rate of increase for blue whales in the central North Atlantic to be 9% annually (3.83 SE) between 1987 and 2001.

North Pacific. Estimates of blue whale abundance are uncertain. Prior to whaling, Gambell (1976) reported there may have been as many as 4,900 blue whales. Blue whales were hunted in the Pacific Ocean, where 5,761 were killed from 1889–1965 (Perry et al. 1999). The IWC banned commercial whaling in the North Pacific in 1966, although Soviet whaling continued after the ban. In the eastern North Pacific, the minimum stock abundance (based upon surveys in U.S. EEZ waters) is thought to be 1,384 whales, but no minimum estimate has been established (Carretta et al. 2006). Although blue whale abundance has likely increased since its protection in 1966, the possibility of unauthorized harvest by Soviet whaling vessels, incidental ship strikes, and gillnet mortalities make this uncertain. Punt (2010) estimated the rate of increase for blue whales in the eastern North Pacific to be 3.2% annually (1.4 SE) between 1991 and 2005.

Calambokidis and Barlow (2004) estimated roughly 3,000 blue whales inhabit waters off California, Oregon, and Washington based on line-transect surveys and 2,000 based on capture-recapture methods. Carretta et al.(2006) noted that the best estimate of abundance off California, Oregon, and Washington is an average of line-transect and capture-recapture estimates (1,744). Barlow (2003) reported mean group sizes of 1.0–1.9 during surveys off California, Oregon, and Washington. A density estimate of 0.0003 individuals/km^2 was given for waters off Oregon/Washington, and densities off California ranged from 0.001-0.0033 individuals/km^2 (Barlow 2003).

Southern Hemisphere. Estimates of 4-5% for an average rate of population growth have been proposed (Yochem and Leatherwood 1985). However, a recent estimate of population growth for Antarctic blue whales throughout the region was 7.3% (Branch et al. 2007). Punt (2010) estimated the rate of increase for blue whales in the Southern Hemisphere to be 8.2% annually (3.37 SE) between 1978 and 2004. Branch et al. (2007) also included an estimate of 1,700 individuals south of 60º. Antarctic blue whales remain severely depleted with the 1996 estimate only 0.7% of pre-whaling levels (IWC 2005). Blue whales along Chile have been estimated to number between 7 and 9% of historical abundance (Williams et al. 2011).

Blue whales were the mainstay of whaling in the region once the explosive harpoon was developed in the late nineteenth century (Shirihai 2002). During the early 1900s, the species became a principal target of the whaling industry throughout the world, with the majority killed in the Southern Hemisphere. Approximately 330,000–360,000 blue whales were harvested from 1904 to 1967 in the Antarctic alone, reducing their abundance to <3% of their original numbers (Perry et al. 1999; Reeves et al. 2003b). Blue whales were protected in portions of the Southern Hemisphere beginning in 1939, and received full protection in the Antarctic in 1966.

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

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

Ship strike is presently a concern for blue whale recovery. Ship strikes have recently averaged roughly one every other year (eight ship strike incidents are known Jensen and Silber (2004), but in September 2007, ships struck five blue whales within a few-day period off southern California (Calambokidis pers. comm. 2008). Dive data support a surface-oriented behavior during nighttime that would make blue whales particularly vulnerable to ship strikes. There are concerns that, like right whales, blue whales may surface when approached by large vessels; a behavior that would increase their likelihood of being struck. Protective measures are not currently in place. In the California/Mexico stock, annual incidental mortality due to ship strikes averaged one whale every 5 years, but we cannot determine if this reflects the actual number of blue whales struck and killed by ships (i.e., individuals not observed when struck and those who do not strand; Barlow et al. (1997). It is believed that the vast majority of ship strike mortalities are never identified, and that actual mortality is higher than currently documented.

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

There is a paucity of contaminant data regarding blue whales. Available information indicates that organochlorines, including dichloro-diphenyl-trichloroethane (DDT), polychlorinated biphenyls (PCB), benzene hexachloride (HCH), hexachlorobenzene (HCB), chlordane, dieldrin, methoxychlor, and mirex have been isolated from blue whale blubber and liver samples (Gauthier et al. 1997b; Metcalfe et al. 2004). Contaminants transfer between mother and calf meaning that young often start life with concentrations of contaminants equal to their mothers, before accumulating additional contaminant loads during life and passing higher loads to the next generation (Gauthier et al. 1997a; Metcalfe et al. 2004).

Critical habitat. The NMFS has not designated critical habitat for blue whales.

Fin whale

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

Subspecies. There are two recognized subspecies of fin whales, *Balaenoptera physalus physalus*, which occurs in the North Atlantic Ocean, and *B. p. quoyi*, which occurs in the Southern Ocean. These subspecies and North Pacific fin whales appear to be organized into separate populations, although there is a lack of consensus in the published literature as to population structure.

Population structure. Population structure has undergone only a rudimentary framing. Genetic studies by Bérubé et al. (1998) indicate that there are significant genetic differences among fin whales in differing geographic areas (Sea of Cortez, Gulf of St. Lawrence, and Gulf of Maine). Further, individuals in the Sea of Cortez may represent an isolated population from other eastern North Pacific fin whales (Berube et al. 2002). Even so, mark-recapture studies also demonstrate that individual fin whales migrate between management units designated by the IWC (Mitchell 1974b; Sigujónsson and Gunnlaugsson 1989).

North Atlantic. Fin whales are common off the Atlantic coast of the U.S. in waters immediately off the coast seaward to the continental shelf (about the 1,800 m contour).

Fin whales occur during the summer from Baffin Bay to near Spitsbergen and the Barents Sea, south to Cape Hatteras in North Carolina and off the coasts of Portugal and Spain (Rice 1998a). In areas north of Cape Hatteras, fin whales account for about 46% of the large whales observed in 1978-1982 surveys (CETAP 1982). Little is known about the winter habitat of fin whales, but in the western North Atlantic, the species has been found from Newfoundland south to the Gulf of Mexico and Greater Antilles, and in the eastern North Atlantic their winter range extends from the Faroes and Norway south to the Canary Islands. Fin whales in the eastern North Atlantic have been found in highest densities in the Irminger Sea between Iceland and Greenland (Víkingsson et al. 2009). The singing location of fin whales in the Davis Strait and Greenland has been correlated with sea ice fronts; climate change may impact fin whale distribution and movement by altering sea ice conditions (Simon et al. 2010). A general fall migration from the Labrador and Newfoundland region, south past Bermuda, and into the West Indies has been theorized (Clark 1995). Historically, fin whales were by far the most common large whale found off Portugal (Brito et al. 2009).

Fin whales are also endemic to the Mediterranean Sea, where (at least in the western Mediterranean), individuals tend to aggregate during summer and disperse in winter over large spatial scales (Cotte et al. 2009). Mediterranean fin whales are genetically distinct from fin whales in the rest of the North Atlantic at the population level (Berube et al. 1999). However, some fin whales from the northeastern North Atlantic have been tracked into the Mediterranean during winter and overlap in time and space with the Mediterranean population may exist (Castellote et al. 2010). Individuals also tend to associate with colder, saltier water, where steep changes in temperature, and where higher northern krill densities would be expected (Cotte et al. 2009). A genetically distinct population resides year-round in the Ligurian Sea (IWC 2006a).

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

(Mizroch et al. 1999b). A resident fin whale population may exist in the Gulf of California (Tershy et al. 1993).

Fin whales are observed year-round off central and southern California with peak numbers in the summer and fall (Barlow 1997a; Dohl et al. 1983; Forney et al. 1995). Peak numbers are seen during the summer off Oregon, and in summer and fall in the Gulf of Alaska and southeastern Bering Sea (Moore et al. 2000; Perry et al. 1999). Fin whales are observed feeding in Hawaiian waters during mid-May, and their sounds have been recorded there during the autumn and winter (Balcomb 1987; Northrop et al. 1968; Shallenberger 1981; Thompson and Friedl 1982). Fin whales in the western Pacific winter in the Sea of Japan, the East China, Yellow, and Philippine seas (Gambell 1985a).

Southern Hemisphere. Fin whales range from near 40° S (Brazil, Madagascar, western Australia, New Zealand, Colombia, Peru, and Chile) during the austral winter southward to Antarctica (Rice 1998a). Fin whales appear to be present in Antarctic waters only from February-July and were not detected in the Ross Sea during year-round acoustic surveys (Sirovic et al. 2009). Fin whales in the action area likely would be from the New Zealand stock, which summers from 170° E to 145° W and winters in the Fiji Sea and adjacent waters (Gambell 1985a).

Age distribution. Aguilar and Lockyer (1987) suggested annual natural mortality rates in northeast Atlantic fin whales may range from 0.04 to 0.06. Fin whales live 70-80 years (Kjeld et al. 2006).

Reproduction. Fin whales reach sexual maturity between 5-15 years of age (COSEWIC 2005; Gambell 1985a; Lockyer 1972). Mating and calving occurs primarily from October-January, gestation lasts ~11 months, and nursing occurs for 6-11 months (Boyd et al. 1999; Hain et al. 1992). The average calving interval in the North Atlantic is estimated at about 2-3 years (Agler et al. 1993; Christensen et al. 1992a). The location of winter breeding grounds is uncertain but mating is assumed to occur in pelagic mid-latitude waters (Perry et al. 1999). This was recently contradicted by acoustic surveys in the Davis Strait and off Greenland, where singing by fin whales peaked in November through December; the authors suggested that mating may occur prior to southbound migration (Simon et al. 2010). Although seasonal migration occurs between presumed foraging and breeding locations, fin whales have been acoustically detected throughout the North Atlantic Ocean and Mediterranean Sea year-round, implying that not all individuals follow a set migratory pattern (Notarbartolo-Di-Sciara et al. 1999; Simon et al. 2010).

Feeding. Fin whales in the North Atlantic eat pelagic crustaceans (mainly krill and schooling fish such as capelin, herring, and sand lance (Borobia and Béland 1995; Christensen et al. 1992a; Hjort and Ruud 1929; Ingebrigtsen 1929; Jonsgård 1966; Mitchell 1974b; Overholtz and Nicolas 1979; Sergeant 1977; Shirihai 2002; Watkins et al. 1984). In the North Pacific, fin whales also prefer euphausiids and large copepods, followed by schooling fish such as herring, walleye pollock, and capelin (Kawamura 1982a; Kawamura 1982b; Ladrón De Guevara et al. 2008; Nemoto 1970; Paloma et al. 2008). Fin whales frequently forage along cold eastern current boundaries (Perry et al. 1999). Antarctic fin whales feed on krill, *Euphausia superba*, which occurs in dense near-surface schools (Nemoto 1959). However, off the coast of Chile, fin whales are known to feed on the euphausiid *E. mucronata* (Antezana 1970; Perez et al. 2006). Feeding may occur in waters as shallow as 10 m when prey are at the surface, but most foraging is observed in high-productivity, upwelling, or thermal front marine waters (Gaskin 1972; Nature

Conservancy Council 1979 as cited in ONR 2001; Panigada et al. 2008; Sergeant 1977). While foraging, fin whales in the Mediterranean Sea have been found to move through restricted territories in a convoluted manner (Lafortuna et al. 1999).

Diving. The amount of time fin whales spend at the surface varies. Some authors have reported that fin whales make 5-20 shallow dives, each of 13-20 s duration, followed by a deep dive of 1.5-15 min (Gambell 1985a; Lafortuna et al. 2003; Stone et al. 1992). Other authors have reported that the fin whale's most common dives last 2-6 min (Hain et al. 1992; Watkins 1981). The most recent data support average dives of 98 m and 6.3 min for foraging fin whales, while non-foraging dives are 59 m and 4.2 min (Croll et al. 2001). However, Lafortuna et al. (1999) found that foraging fin whales have a higher blow rate than when traveling. Foraging dives in excess of 150 m are known (Panigada et al. 1999). In waters off the U.S. Atlantic Coast, individuals or duos represented about 75% of sightings (Hain et al. 1992). Individuals or groups of less than five individuals represented about 90% of observations. Barlow (2003) reported mean group sizes of 1.1–4.0 during surveys off California, Oregon, and Washington.

Vocalization and hearing. Fin whales produce a variety of low-frequency sounds in the 10-200 Hz range (Edds 1988; Thompson et al. 1992; Watkins 1981; Watkins et al. 1987). Typical vocalizations are long, patterned pulses of short duration (0.5-2 s) in the 18-35 Hz range, but only males are known to produce these (Croll et al. 2002; Patterson and Hamilton 1964). Richardson et al. (1995b) reported the most common sound as a 1 s vocalization of about 20 Hz, occurring in short series during spring, summer, and fall, and in repeated stereotyped patterns during winter. Au (2000) reported moans of 14-118 Hz, with a dominant frequency of 20 Hz, tonal vocalizations of 34-150 Hz, and songs of 17-25 Hz (Cummings and Thompson 1994; Edds 1988; Watkins 1981). Source levels for fin whale vocalizations are 140-200 dB re 1μPa·m (Clark and Ellison. 2004; Erbe 2002b). The source depth of calling fin whales has been reported to be about 50 m (Watkins et al. 1987).

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

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

Status and trends. Fin whales were originally listed as endangered in 1970 (35 FR 18319), and this status continues since the inception of the ESA in 1973. Although fin whale population structure remains unclear, various abundance estimates are available (Table 4). Pre-exploitation fin whale abundance is estimated at 464,000 individuals worldwide; the estimate for 1991 was roughly 25% of this (Braham 1991). Historically, worldwide populations were severely depleted by commercial whaling, with more than 700,000 whales harvested in the twentieth century (Cherfas 1989).

North Atlantic. 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. However, over 48,000 fin whales were caught between 1860-1970 (Braham 1991). Although protected by the IWC, from 1988-1995 there have been 239 fin whales harvested from the North Atlantic. Recently, Iceland resumed whaling of fin whales despite the 1985 moratorium imposed by the IWC. Forcada et al. (1996) estimated that 3,583 individuals (95% CI = 2,130- 6,027) inhabit the western Mediterranean Sea. Goujon et al. (1994) estimated 7,000-8,000 fin whales in the Bay of Biscay. Vikingsson et al. (2009) estimated roughly 20,000 fin whales to be present in a large portion of the eastern North Atlantic in 1995, which increased to roughly 25,000 in 2001. The authors concluded that actual numbers were likely higher due to negative bias in their analysis, and that the population(s) were increasing at 4% annually (Víkingsson et al. 2009). The abundance of fin whales in the Baffin Bay-Davis Strait summer feeding area is believed to be increasing (Heide-Jorgensen et al. 2010).

Table 4. Summary of past and present fin whale abundance.

Region	Population, stock, or study area	Pre-exploitation estimate	95% C.I.	Current estimate	95% C.I.	Source
Global	--	>464,000	--	119,000	--	(Braham 1991)
North Atlantic	Basinwide	30,000-50,000	--	--	--	(Sergeant 1977)
		360,000	249,000-481,000	--	--	(Roman and Palumbi 2003)
	Central and Northeastern Atlantic	--	--	30,000	23,000-39,000	(IWC 2007)
	Western North Atlantic	--	--	3,590-6,300	--	(Braham 1991)
	NMFS-Western North Atlantic stock	--	--	2,269	CV=0.37	(Angliss and Outlaw 2009)
	Northeastern U.S. Atlantic continental shelf	--	--	2,200-5,000	--	(Hain et al. 1992; Waring et al. 2000)
	IWC-Newfoundland-Labrador stock	--	--	13,253	0-50,139*	(IWC 1992)
	IWC-British Isles, Spain, and Portugal stock	10,500	9,600-11,400	4,485	3,369-5,600	(Braham 1991)
		--	--	17,355	10,400-28,900	(Buckland et al. 1992)
	IWC-east Greenland-Iceland stock	--	--	11,563	5,648-17,478*	(Gunnlaugsson and Sigurjónsson 1990)
	IWC-west	--	--	1,700	840-3,500	(IWC 2006a)

Region	Population, stock, or study area	Pre-exploitation estimate	95% C.I.	Current estimate	95% C.I.	Source
	Greenland stock					
North Pacific	Basinwide	42,000-45,000	--	16,625	14,620-18,630	(Braham 1991; Ohsumi and Wada 1974)
	Central Bering Sea	--	--	4,951	2,833-8,653	(Moore et al. 2002)
	NMFS – northeast Pacific stock-west of Kenai Peninsula	--	--	5,700	--	(Angliss and Outlaw 2009)
	NMFS-California/ Oregon/Washington stock	--	--	2,636	CV=0.15	(Carretta et al. 2008)
	NMFS-Hawaii stock	--	--	174	0-420*	(Carretta et al. 2008)
	Sea of Okhotsk	--	--	13,000	6,700-25,600	Miyashita and Kato (2005)
Southern Hemisphere	Basinwide	400,000	--	85,200	--	(Braham 1991; IWC 1979)
	South of 60^{o}S	--	--	1,735	514-2,956	(IWC 1996)
	South of 30^{o}S	--	--	15,178	--	(IWC 1996)
	Scotia Sea and Antarctic Peninsula	--	--	4,672	792-8,552	(Hedley et al. 2001; Reilly et al. 2004)

*Note: Confidence Intervals (C.I.) not provided by the authors were calculated from Coefficients of Variation (C.V.) where available, using the computation from Gotelli and Ellison (2004).

North Pacific. The status and trend of fin whale populations is largely unknown. Over 26,000 fin whales were harvested between 1914-1975 (Braham 1991 as cited in Perry et al. 1999). NMFS estimates roughly 3,000 individuals occur off California, Oregon, and Washington based on ship surveys in summer/autumn of 1996, 2001, and 2005, of which estimates of 283 and 380 have been made for Oregon and Washington alone (Barlow 2003; Barlow and Taylor 2001; Forney 2007). Punt (2010) estimated the rate of increase for fin whales in the eastern North Pacific to be 4.8% annually (3.24 SE) between 1987 and 2003.

Southern Hemisphere. The Southern Hemisphere population was one of the most heavily exploited whale populations under commercial whaling. From 1904 to 1975, over 700,000 fin whales were killed in Antarctic whaling operations (IWC 1990). Harvests increased substantially upon the introduction of factory whaling ships in 1925, with an average of 25,000 caught annually from 1953-1961 (Perry et al. 1999). Current estimates are a tiny fraction of former abundance.

Natural threats. Natural sources and rates of mortality are largely unknown, but Aguilar and Lockyer (1987) suggested annual natural mortality rates might range from 0.04 to 0.06 for northeast Atlantic fin whales. The occurrence of the nematode *Crassicauda boopis* appears to increase the potential for kidney failure and may be preventing some fin whale populations from recovering (Lambertsen 1992). Adult fin whales engage in a flight responses (up to 40 km/h) to evade killer whales, which involves high energetic output, but show little resistance if overtaken (Ford and Reeves 2008). Shark attacks may also result in serious injury or death in very young and sick individuals (Perry et al. 1999).

Anthropogenic threats. Fin whales have undergone significant exploitation, but are currently protected under the IWC. Fin whales are still hunted in subsistence fisheries off West Greenland. In 2004, five males and six females were killed, and two other fin whales were struck and lost. In 2003, two males and four females were landed and two others were struck and lost (IWC 2005). Between 2003 and 2007, the IWC set a catch limit of up to 19 fin whales in this subsistence fishery. However, the scientific recommendation was to limit the number killed to four individuals until accurate populations could be produced (IWC 2005). In the Antarctic Ocean, fin whales are hunted by Japanese whalers who have been allowed to kill up to 10 fin whales each ear for the 2005-2006 and 2006-2007 seasons under an Antarctic Special Permit NMFS (2006b). The Japanese whalers plan to kill 50 whales per year starting in the 2007-2008 season and continuing for the next 12 years (IWC 2006b; Nishiwaki et al. 2006).

Fin whales experience significant injury and mortality from fishing gear and ship strikes (Carretta et al. 2007a; Douglas et al. 2008; Lien 1994; Perkins and Beamish 1979; Waring et al. 2007). Between 1969-1990, 14 fin whales were captured in coastal fisheries off Newfoundland and Labrador; of these seven are known to have died because of capture (Lien 1994; Perkins and Beamish 1979). In 1999, one fin whale was reported killed in the Gulf of Alaska pollock trawl fishery and one was killed the same year in the offshore drift gillnet fishery (Angliss and Outlaw 2005; Carretta et al. 2004). According to Waring et al. (2007), four fin whales in the western North Atlantic died or were seriously injured in fishing gear, while another five were killed or injured as a result of ship strikes between January 2000 and December 2004. Jensen and Silber (2004) review of the NMFS' ship strike database revealed fin whales as the most frequently confirmed victims of ship strikes (26% of the recorded ship strikes [n = 75/292 records]), with most collisions occurring off the east coast, followed by the west coast of the U.S. and Alaska/Hawaii. Between 1999-2005, there were 15 reports of fin whales strikes by vessels along the U.S. and Canadian Atlantic coasts (Cole et al. 2005a; Nelson et al. 2007a). Of these, 13 were confirmed, resulting in the deaths of 11 individuals. Five of seven fin whales stranded along Washington State and Oregon showed evidence of ship strike with incidence increasing since 2002 (Douglas et al. 2008). Similarly, 2.4% of living fin whales from the Mediterranean show ship strike injury and 16% of stranded individuals were killed by vessel collision (Panigada et al. 2006). There are also numerous reports of ship strikes off the Atlantic coasts of France and England (Jensen and Silber 2004).

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

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

The organochlorines DDE, DDT, and PCBs have been identified from fin whale blubber, but levels are lower than in toothed whales due to the lower level in the food chain that fin whales feed at (Aguilar and Borrell 1988; Borrell 1993; Borrell and Aguilar 1987; Henry and Best 1983; Marsili and Focardi 1996). Females contained lower burdens than males, likely due to mobilization of contaminants during pregnancy and lactation (Aguilar and Borrell 1988; Gauthier et al. 1997a; Gauthier et al. 1997b). Contaminant levels increase steadily with age until sexual maturity, at which time levels begin to drop in females and continue to increase in males(Aguilar and Borrell 1988).

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

Critical habitat. The NMFS has not designated critical habitat for fin whales.

Humpback whale

Description of the species. Humpback whales are a cosmopolitan species that occur in the Atlantic, Indian, Pacific, and Southern oceans. Humpback whales migrate seasonally between warmer, tropical or sub-tropical waters in winter months (where they breed and give birth to calves, although feeding occasionally occurs) and cooler, temperate or sub-Arctic waters in summer months (where they feed) (Gendron and Urban 1993). In both regions, humpback whales tend to occupy shallow, coastal waters. However, migrations are undertaken through deep, pelagic waters (Winn and Reichley 1985).

Population designations. Populations have been relatively well defined for humpback whales

North Atlantic. Humpback whales range from the mid-Atlantic bight and the Gulf of Maine across the southern coast of Greenland and Iceland to Norway in the Barents Sea. Whales migrate to the western coast of Africa and the Caribbean Sea during the winter. Humpback whales aggregate in four summer feeding areas: Gulf of Maine and eastern Canada, west Greenland, Iceland, and Norway (Boye et al. 2010; Katona and Beard 1990; Smith et al. 1999).

Increasing range and occurrence in the Mediterranean Sea coincides with population growth and may represent reclaimed habitat from pre-commercial whaling (Frantzis et al. 2004; Genov et al. 2009). The principal breeding range for Atlantic humpback whales lies from the Antilles and northern Venezuela to Cuba (Balcomb III and Nichols 1982; Whitehead and Moore 1982; Winn et al. 1975). The largest breeding aggregations occur off the Greater Antilles where humpback whales from all North Atlantic feeding areas have been photo-identified (Clapham et al. 1993; Katona and Beard 1990; Mattila et al. 1994; Palsbøll et al. 1997; Smith et al. 1999; Stevick et al. 2003b). However, the possibility of historic and present breeding further north remains enigmatic but plausible (Smith and G.Pike 2009). Winter aggregations also occur at the Cape Verde Islands in the eastern North Atlantic and along Angola (Cerchio et al. 2010b; Reeves et al. 2002; Reiner et al. 1996; Weir 2007). Accessory and historical aggregations also occur in the eastern Caribbean (Levenson and Leapley 1978; Mitchell and Reeves 1983; Reeves et al. 2001a; Reeves et al. 2001b; Schwartz 2003; Smith and Reeves 2003; Swartz et al. 2003; Winn et al. 1975). To further highlight the "open" structure of humpback whales, a humpback whale migrated from the Indian Ocean to the South Atlantic Ocean, demonstrating that interoceanic movements can occur (Pomilla and Rosenbaum 2005). Genetic exchange at low-latitude breeding groups between Northern and Southern Hemisphere individuals and wider-range movements by males has been suggested to explain observed global gene flow (Rizzo and Schulte 2009). However, there is little genetic support for wide-scale interchange of individuals between ocean basins or across the equator.

North Pacific. Based on genetic and photo-identification studies, the NMFS currently recognizes four stocks, likely corresponding to populations, of humpback whales in the North Pacific Ocean: two in the eastern North Pacific, one in the central North Pacific, and one in the western Pacific (Hill and DeMaster 1998b). Gene flow between them may exist. Humpback whales summer in coastal and inland waters from Point Conception, California, north to the Gulf of Alaska and the Bering Sea, and west along the Aleutian Islands to the Kamchatka Peninsula and into the Sea of Okhotsk (Johnson and Wolman 1984; Nemoto 1957; Tomilin 1967). These whales migrate to Hawaii, southern Japan, the Mariana Islands, and Mexico during winter. However, more northerly penetrations in Arctic waters occur on occasion (Hashagen et al. 2009). The central North Pacific population winters in the waters around Hawaii while the eastern North Pacific population (also called the California-Oregon-Washington-Mexico stock) winters along Central America and Mexico. However, Calambokidis et al. (1997) identified individuals from several populations wintering (and potentially breeding) in the areas of other populations, highlighting the potential fluidity of population structure. Humpback whales were recently found to migrate to the northwestern Hawaiian Islands, where singing has been recorded; this may represent an as yet undescribed breeding group, or expansion of breeding from the main Hawaiian Islands (Lammers et al. 2011). Herman (1979) presented extensive evidence that humpback whales associated with the main Hawaiian Islands immigrated there only in the past 200 years. Winn and Reichley (1985) identified genetic exchange between the humpback whales that winter off Hawaii and Mexico (with further mixing on feeding areas in Alaska) and suggested that humpback whales that winter in Hawaii may have emigrated from Mexican wintering areas. A “population” of humpback whales winters in the South China Sea east through the Philippines, Ryukyu Retto, Ogasawara Gunto, Mariana Islands, and Marshall Islands, with occurrence in the Mariana Islands, Guam, Rota, and Saipan from January-March (Darling and Mori 1993; Eldredge 1991; Eldredge 2003; Rice 1998a). During summer, whales from this population migrate to the Kuril Islands, Bering Sea, Aleutian Islands, Kodiak, Southeast Alaska, and British Columbia to feed (Angliss and Outlaw 2007b; Calambokidis 1997; Calambokidis et al. 2001b).

Arabian Sea. 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 and movements of this group are poorly known (Mikhalev 1997; Rasmussen et al. 2007). Areas of the Mozambique Channel appear to be significant calving and wintering areas for humpback whales (Kiszka et al. 2010).

Southern Hemisphere. Eight proposed stocks, or populations, of humpback whales occur in waters off Antarctica (Figure 1). Individuals from these stocks winter and breed in separate areas and are known to return to the same areas. However, the degree (if any) of gene flow (i.e., adult individuals wintering in different breeding locations) is uncertain (Carvalho et al. 2011). Individuals from breeding grounds in Ecuador are somewhat heterogeneous from individuals in other breeding areas, but appear to maintain a genetic linkage (Felix et al. 2009). Based upon recent satellite telemetry, a revision of stocks A and G may be warranted to reflect stock movements within and between feeding areas separated east of 50º W (Dalla Rosa et al.

2008). In addition to being a breeding area, the west coast of South Africa also appears to serve as a foraging ground due to upwelling of the Benguela Current (Barendse et al. 2010). Females appear in this area in large numbers well before their male counterparts, frequently accompanied by calves (Barendse et al. 2010). Female movement between breeding locations across years has been documented, bringing into question the genetic discreteness of at least Southern Hemisphere populations (Stevick et al. 2011). However, mixing between some populations has not been found (such as between B2 and C1 groups). Sao Tome appears to be primarily a resting, nursing, and calving area with very little breeding occurring (Carvalho et al. 2011).

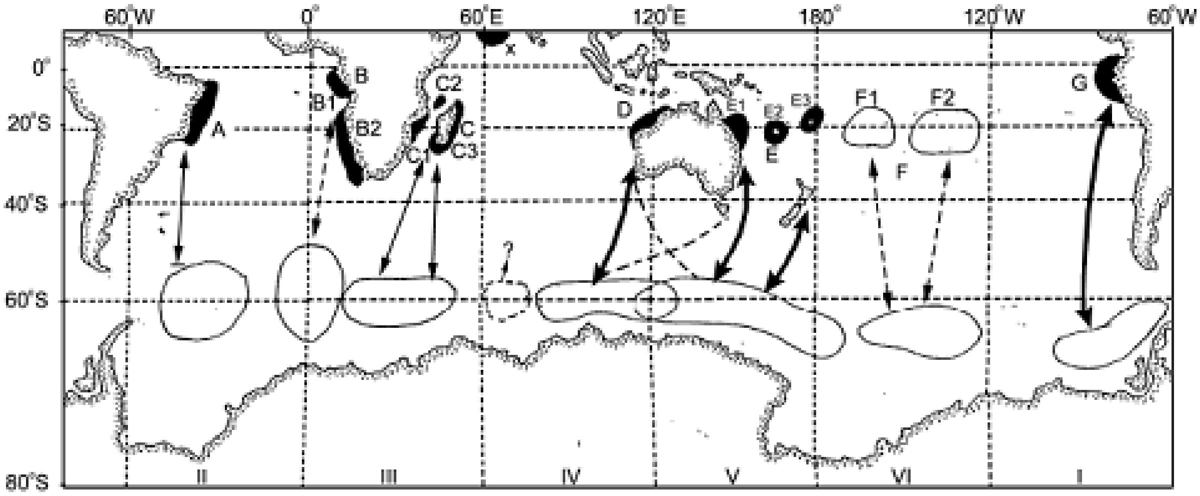

Figure 4. Southern Hemisphere humpback stocks (populations) (IWC 2005).

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

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

Diving. In Hawaiian waters, humpback whales remain almost exclusively within the 1,800 m isobath and usually within water depths of less than 182 m. Maximum diving depths are approximately 170 m (but usually <60 m), with a very deep dive (240 m) recorded off Bermuda (Hamilton et al. 1997). Dives can last for up to 21 min, although feeding dives ranged from 2.1-5.1 min in the North Atlantic (Dolphin 1987). In southeast Alaska, average dive times were 2.8 min for feeding whales, 3.0 min for non-feeding whales, and 4.3 min for resting whales (Dolphin 1987). In the Gulf of California, humpback whale dive durations averaged 3.5 min (Strong 1990). Because most humpback prey is likely found within 300 m of the surface, most humpback dives are probably relatively shallow. In Alaska, capelin are the primary prey of humpback and are found primarily between 92 and 120 m; depths to which humpbacks apparently dive for foraging (Witteveen et al. 2008).

Feeding. During the feeding season, humpback whales form small groups that occasionally aggregate on concentrations of food that may be stable for long-periods of times. Humpbacks use a wide variety of behaviors to feed on various small, schooling prey including krill and fish (Hain et al. 1982; Hain et al. 1995; Jurasz and Jurasz 1979; Weinrich et al. 1992; Witteveen et al. 2011). The principal fish prey in the western North Atlantic are sand lance, herring, and capelin (Kenney et al. 1985). There is good evidence of some territoriality on feeding and calving areas (Clapham 1994; Clapham 1996; Tyack 1981). Humpback whales are generally believed to fast while migrating and on breeding grounds, but some individuals apparently feed while in low-latitude waters normally believed to be used exclusively for reproduction and calf-rearing (Danilewicz et al. 2009; Pinto De Sa Alves et al. 2009). Some individuals, such as juveniles, may not undertake migrations at all (Findlay and Best. 1995). Additional evidence, such as songs sung in northern latitudes during winter, provide additional support to plastic seasonal distribution (Smith and G.Pike 2009). Relatively high rates of resighting in foraging sites in Greenland suggest whales return to the same areas year after year (Kragh Boye et al. 2010).

Vocalization and hearing. We understand humpback whale vocalization much better than we do hearing. Humpback whales produce different sounds that correspond to different functions: feeding, breeding, and other social calls. Males sing complex sounds while in low-latitude breeding areas in a frequency range of 20 Hz to 4 kHz with estimated source levels from 144-174 dB (Au 2000; Au et al. 2006; Frazer and Mercado 2000; Payne 1970; Richardson et al. 1995c; Winn et al. 1970). Males also produce sounds associated with aggression, which are generally characterized as frequencies between 50 Hz to 10 kHz and having most energy below 3 kHz (Silber 1986; Tyack 1983). Such sounds can be heard up to 9 km away (Tyack and Whitehead 1983). Other social sounds from 50 Hz to 10 kHz (most energy below 3 kHz) are also produced in breeding areas (Richardson et al. 1995c; Tyack and Whitehead 1983). While in northern feeding areas, both sexes vocalize in grunts (25 Hz to 1.9 kHz), pulses (25-89 Hz), and songs (ranging from 30 Hz to 8 kHz but dominant frequencies of 120 Hz to 4 kHz) which can be very loud (175-192 dB re 1 μPa at 1 m; (Au 2000; Erbe 2002a; Payne and Payne 1985; Richardson et al. 1995c; Thompson et al. 1986). Humpbacks tend to be less vocal in northern feeding areas than in southern breeding areas (Richardson et al. 1995c). Recently, humpback whales were reported to use echolocation-type clicks that were associated with feeding (Stimpert et al. 2009). The authors suggest that a primitive echoranging capability may exist.

Status and trends. Humpback whales were originally listed as endangered in 1970 (35 FR 18319), and this status remains under the ESA. (Winn and Reichley 1985) argued that the global humpback whale population consisted of at least 150,000 whales in the early 1900s, mostly in the Southern Ocean. In 1987, the global population of humpback whales was estimated at about

10,000 (NMFS 1987). Although this estimate is outdated, it appears that humpback whale numbers are increasing. Table 4 provides estimates of historic and current abundance for ocean regions.

Table 5. Summary of past and present humpback whale abundance.

Region	Population, stock, or study area	Pre-exploitation estimate	95% C.I.	Current estimate	95% C.I.	Source
Global	--	1,000,000	--	--	--	(Roman and Palumbi 2003)
North Atlantic	Basinwide	240,000	156,000-401,000*	11,570	10,005-13,135*	(Roman and Palumbi 2003) (Stevick et al. 2001) *in* (Waring et al. 2004)
	Basinwide - Females	--	--	2,804	1,776-4,463	(Palsbøll et al. 1997)
	Basinwide - Males	--	--	4,894	3,374-7,123	(Palsbøll et al. 1997)
	Western North Atlantic from Davis Strait, Iceland to the West Indies	>4,685*	--	--	--	*circa 1865; (Mitchell and Reeves 1983)
	NMFS - Gulf of Maine stock	--	--	847	CV=0.55	(Waring et al. 2009)
	NMFS - Gulf of Maine stock, including a portion of Scotian Shelf	--	--	902	177-1,627*	(Clapham et al. 2003)
	Northeast Atlantic - Barents and Norwegian Seas	--	--	889	331-1,447*	(Øien 2001) *in* (Waring et al. 2004)
North Pacific	Basinwide	15,000	--	6,000-8,000	--	(Calambokidis et al. 1997)
	NMFS - Western North Pacific stock	--	--	394	329-459*	(Angliss and Allen 2007)
	NMFS - Central North Pacific stock	--	--	5,833	CV=0.30	(Angliss and Outlaw 2009)
	NMFS - Eastern North Pacific stock	--	--	1,391	1,331-1,451*	(Carretta et al. 2009)
Indian Ocean	Arabian Sea	--	--	56	35-255	Minton et al. (2003) *in* (Bannister 2005)
Southern Hemisphere	Basinwide	100,000	--	19,851	--	(Gambell 1976; IWC 1996)
	South of 60^{o}S	--	--	4,660	2,897-6,423	(IWC 1996)

*Note: Confidence Intervals (C.I.) not provided by the authors were calculated from Coefficients of Variation (C.V.) where available, using the computation from Gotelli and Ellison (2004).

North Atlantic. The best available estimate of North Atlantic abundance comes from 1992-1993 mark-recapture data, which generated an estimate of 11,570 humpback whales (Stevick et al. 2003a). Historical estimates have ranged from 40,000-250,000 (Smith and G.Pike 2009). Smith and Reeves (2010) estimated that roughly 31,000 individuals were removed from the North Atlantic due to whaling since the 1600s. Estimates of animals on Caribbean breeding grounds exceed 2,000 individuals (Balcomb III and Nichols 1982). Several researchers report an increasing trend in abundance for the North Atlantic population, which is supported by increased sightings within the Gulf of Maine feeding aggregation (Barlow 1997b; Katona and Beard 1990; Smith et al. 1999; Waring et al. 2001). The rate of increase varies from 3.2-9.4%, with rates of increase slowing over the past two decades (Barlow 1997b; Katona and Beard 1990; Stevick et al. 2003a). If the North Atlantic population has grown according to the estimated instantaneous rate of increase ($r = 0.0311$), this would lead to an estimated 18,400 individual whales in 2008 (Stevick et al. 2003a). Punt (2010) estimated the rate of increase for humpback whales in the

Gulf of Maine to be 6.3% annually (1.2 SE). Pike et al. (2009a) suggested that the eastern and northeastern waters off Iceland are areas of significant humpback utilization for feeding, estimating nearly 5,000 whales in 2001 and proposing an annual growth rate of 12% for the area. The authors suggest that humpback whales in the area had probably recovered from whaling. However, recent data suggest that the upward growth may have slowed or ceased around Iceland according to analysis of survey data there (Pike et al. 2010).

North Pacific. The pre-exploitation population size may have been as many as 15,000 humpback whales, and current estimates are 6,000-8,000 whales (Calambokidis et al. 1997; Rice 1978a). It is estimated that 15,000 humpback whales resided in the North Pacific in 1905 (Rice 1978a). However, from 1905 to 1965, nearly 28,000 humpback whales were harvested in whaling operations, reducing the number of all North Pacific humpback whale to roughly 1,000 (Perry et al. 1999). Estimates have risen over time from 1,407-2,100 in the 1980s to 6,010 in 1997 (Baker 1985; Baker and Herman 1987; Calambokidis et al. 1997; Darling and Morowitz 1986). Because estimates vary by methodology, they are not directly comparable and it is not clear which of these estimates is more accurate or if the change from 1,407 to 6,010 is the result of a real increase or an artifact of model assumptions. Tentative estimates of the eastern North Pacific stock suggest an increase of 6-7% annually, but fluctuations have included negative growth in the recent past (Angliss and Outlaw 2005). However, based upon surveys between 2004 and 2006, Calambokidis et al. (2008) estimated that the number of humpback whales in the North Pacific consisted of about 18,300 whales, not counting calves (Barlow et al. (2009) provided a bias-corrected estimate of 20,800 individuals) and the population was growing at 4.9% annually. Almost half of these whales likely occur in wintering areas around the Hawaiian Islands. Punt (2010) estimated the rate of increase for humpback whales in the eastern North Pacific to be 6.4% annually (0.9 SE) between 1992 and 2003 and 10.0% for Hawaii (3.32 SE).

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 1998b). 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 2007a; Calambokidis 1997; Calambokidis et al. 2001a).

Arabian Sea. The population inhabiting the Arabian Sea likely numbers a few hundred individuals at most (Minton et al. 2008). This population likely was much larger prior to exploitation in 1966 by Soviet whaling, with individuals found along not only Oman, but Yemen, Iran, Pakistan, and India (Mikhalev 2000; Minton et al. 2008; Reeves et al. 1991; Slijper et al. 1964; Wray and Martin. 1983).

Southern Hemisphere. The IWC recently compiled population data on humpback whales in the Southern Hemisphere. Approximately 42,000 Southern Hemisphere humpbacks can be found south of 60° S during the austral summer feeding season (IWC 2007). However, humpback whales in this region experienced severe whaling pressure. Based upon whaling logs, particularly by Soviet vessels, at least 75,542 humpback whales were harvested from Antarctic waters from 1946 through 1973, largely from management areas IV, V, and VI (Clapham et al. 2009). One-third of these catches occurred from 1959-1961 in Area V. These numbers support

Southern Hemisphere humpbacks being well below their carrying capacities (Clapham et al. 2009). Recent surveys off the Brazilian breeding grounds suggests a population of 6,404 individuals in this area (Andriolo et al. 2010). Modeling efforts to bound the number of individuals within Oceania have estimated 2,300-3,500 individuals divided amongst various populations/subpopulations (Constantine et al. 2010). A 2009 spike in calf mortality along western Australia brings into question whether carrying capacity has been reached by this population or other factors have increased mortality (Coughran and Gales 2010). Some vital rates of the humpback whale population summering off eastern Australia (E1) were recently estimated, including adult annual survival of 0.925, subadult survival of 0.70 (Hoffman et al. 2010). Growth rates for certain age classes included 10.7% for adult females and 12.4% for juveniles (Hoffman et al. 2010). Punt (2010) estimated the rate of increase for humpback whales off eastern and western Australia to be 10.9 and 10.1% annually, respectively (0.23 and 4.69 SE, respectively).

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

Parasites and biotoxins from red-tide blooms are other potential causes of mortality (Perry et al. 1999). The occurrence of the nematode *Crassicauda boopis* appears to increase the potential for kidney failure in humpback whales and may be preventing some populations from recovering (Lambertsen 1992). Studies of 14 humpback whales that stranded along Cape Cod between November 1987 and January 1988 indicate they apparently died from a toxin produced by dinoflagellates during this period.

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

Humpback whales are also killed or injured during interactions with commercial fishing gear. Like fin whales, fishing gear entangles humpback whales off Newfoundland and Labrador, Canada. A total of 595 humpback whales were reported captured in coastal fisheries in those two provinces between 1969 and 1990, of which 94 died (Lien 1994; Perkins and Beamish 1979). Along the Atlantic coast of the U.S. and the Maritime Provinces of Canada, there were 160 reports of humpback whales being entangled in fishing gear between 1999 and 2005 (Cole et al. 2005b; Nelson et al. 2007b). Of these, 95 entangled humpback whales were confirmed, with 11 whales sustaining injuries and nine dying of their wounds. Several humpback whales are also known to have become entangled in the North Pacific (Angliss and Outlaw 2007b; Hill et al. 1997). Between 1998 and 2005, observers identified 12 humpback whales injured or killed by fisheries off the U.S. west coast (NMFS, unpublished data).

More humpback whales are killed in collisions with ships than any other whale species except fin whales (Jensen and Silber 2003). Along the Pacific coast, a humpback whale is known to be killed about every other year by ship strikes (Barlow et al. 1997). Of 123 humpback whales that stranded along the Atlantic coast of the U.S. between 1975 and 1996, 10 (8.1%) 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. 2005b; Nelson et al. 2007b). Of these reports, 13 were confirmed as ship strikes and in seven cases, ship strike was determined to be the cause of death. In the Bay of Fundy, recommendations for slower vessel speeds to avoid right whale ship strike appear to be largely ignored (Vanderlaan et al. 2008). New rules for seasonal (June through December) slowing of vessel traffic to 10 knots and changing shipping lanes by less than one nautical mile to avoid the greatest concentrations of right whales are expected to reduce the chance of humpback whales being hit by ships by 9%. The first estimate of population-level effects of entanglement were recently produced, with over 12% of the Gulf of Maine population of humpbacks acquiring new scars from entanglement interactions annually (Mattila and Rowles 2010). Along the Pacific U.S. coast, a humpback whale is known to be killed about every other year by ship strikes (Barlow et al. 1997). One sei whale was killed in a collision with a vessel off the coast of Washington in 2003 (Waring et al. 2008). Two whales have been struck offshore of Japan (Jensen and Silber 2003).

Whale watching is a rapidly-growing activity in Japan, with over 45 businesses operating 185 vessels in 30 communities (Hoyt 1993; Hoyt 2001). Humpback whales are targeted in several locations (Hoyt 1993; Hoyt 2001; Mori 1999; Uchida 1997). Peak season is in winter through summer, depending upon location and target species (Hoyt 1993).

Organochlorines, including PCB and DDT, have been identified from humpback whale blubber (Gauthier et al. 1997a). Higher PCB levels have been observed in Atlantic waters versus Pacific waters along the United States and levels tend to increase with individual age (Elfes et al. 2010). Although humpback whales in the Gulf of Maine and off Southern California tend to have the highest PCB concentrations, overall levels are on par with other baleen whales, which are generally lower than odontocete cetaceans (Elfes et al. 2010). As with blue whales, these contaminants are transferred to young through the placenta, leaving newborns with contaminant loads equal to that of mothers before bioaccumulating additional contaminants during life and passing the additional burden to the next generation (Metcalfe et al. 2004). Contaminant levels are relatively high in humpback whales as compared to blue whales. Humpback whales feed higher on the food chain, where prey carry higher contaminant loads than the krill that blue whales feed on.

Critical habitat. The NMFS has not designated critical habitat for humpback whales.

North Pacific right whale

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

Current information on the seasonal distribution of right whales is spotty. In the eastern North Pacific, this includes sightings over the middle shelf of the Bering Sea, Bristol Bay, Aleutian and Pribilof Islands (Goddard and Rugh 1998; Hill and DeMaster 1998a; Perryman et al. 1999; Wade et al. 2006b; Waite et al. 2003). Some more southerly records also record occurrence along Hawaii, California, Washington, and British Columbia (Herman et al. 1980; Scarff 1986a). Records from Mexico and California may suggest historical wintering grounds in offshore southern North Pacific latitudes (Brownell et al. 2001a; Gregr and Coyle. 2009). In the area of the Shatsky Rise, right whales have been spotted to the north of the action area. During the summer and fall time frame, most occurrences have been recorded north of 35˚ N (Matsuoka et al. 2009).

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

Lifespan. Lifespans of up to 70 years can be expected based upon North Atlantic right whale data.

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

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

Historical concentrations of sightings in the Bering Sea together with some recent sightings indicate that this region, together with the Gulf of Alaska, may represent an important summer habitat for eastern North Pacific right whales (Brownell et al. 2001b; Clapham et al. 2004a; Goddard and Rugh 1998; Scarff 1986a; Shelden et al. 2005a). Few sighting data are available from the eastern North Pacific, with a single sighting of 17 individuals in the southeast Bering Sea being by far the greatest known occurrence (Wade et al. 2006a). Some further sightings have occurred in the northern Gulf of Alaska (Wade et al. 2006a). Recent eastern sightings tend to occur over the continental shelf, although acoustic monitoring has identified whales over abyssal waters (Mellinger et al. 2004). It has been suggested that North Pacific right whales have shifted their preferred habitat as a result of reduced population numbers, with oceanic habitat taking on a far smaller component compared to shelf and slope waters (Shelden et al. 2005b). The area where North Pacific right whales are densest in the Gulf of Alaska is between 150 and 170° W and south to 52° N (Shelden and Clapham 2006).

Migration and movement. Historical sighting and catch records provide the only information on possible migration patterns for North Pacific right whales (Omura 1958; Omura et al. 1969a; Scarff 1986a). During summer, whales have been found in the Gulf of Alaska, along both coasts of the Kamchatka Peninsula, the Kuril Islands, the Aleutian Islands, the southeastern Bering Sea, and in the Okhotsk Sea. Fall and spring distribution was the most widely dispersed, with whales occurring in mid-ocean waters and extending from the Sea of Japan to the eastern Bering Sea. In winter, right whales have been found in the Ryukyu Islands (south of Kyushu, Japan), the Bonin Islands, the Yellow Sea, and the Sea of Japan (Clapham et al. 2004a; Omura 1986; Shelden et al. 2005a). Whalers never reported winter calving areas in the North Pacific and where calving occurs remains unknown (Clapham et al. 2004a; Gregr and Coyle. 2009; Scarff 1986a). North Calving grounds may exist in the far offshore Pacific (Scarff 1986b) (Clapham et al. 2004a; Scarff 1991).

Pacific right whales probably migrate north from lower latitudes in spring and may occur throughout the North Pacific from May through August north of 40º N from marginal seas to the Gulf of Alaska and Bering Sea, although absence from the central North Pacific has been argued due to inconsistencies in whaling records (Clapham et al. 2004b; Josephson et al. 2008b). This follows generalized patterns of migration from high-latitude feeding grounds in summer to more temperate, possibly offshore waters, during winter (Braham and Rice 1984; Clapham et al. 2004a; Scarff 1986a).

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

Very little is known about right whales in the eastern North Pacific, which were severely depleted by commercial whaling in the 1800s (Brownell et al. 2001b). At least 11,500 individuals were taken by American whalers in the early- to mid-19th century, but harvesting continued into the 20th century (Best 1987). Illegal Soviet whaling took 372 individuals between 1963 and 1967 (Brownell et al. 2001a). In the last several decades there have been markedly fewer sightings due to a drastic reduction in number, caused by illegal Soviet whaling in the

1960s (Doroshenko 2000). The current population size of right whales in the North Pacific is likely fewer than 1,000 animals compared to possibly 11,000 individuals or more prior to exploitation (NMFS 1991; NMFS 2006d). Wade et al. (2011) used mark-recapture and genotyping methodologies that produced estimates of 31 and 28 individuals, respectively, for individuals in the Bering Sea (likely representing all individuals from the eastern North Pacific).

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

Scientists participating in a recent study utilizing acoustic detection and satellite tracking identified 17 right whales (10 males and 7 females) in the Bearing Sea, which is almost threefold the number seen in any previous year in the last four decades (Wade et al. 2006b). These sightings increased the number of individual North Pacific right whales identified in the genetic catalog for the eastern Bering Sea to 23. Amidst the uncertainty of the eastern North Pacific right whale's future, the discovery of females and calves gives hope that this endangered population may still possess the capacity to recover (Wade et al. 2006b). Available age composition of the North Pacific right whale population indicates most individuals are adults (Kenney 2002). Length measurements for two whales observed off California suggest at least one of these whales was not yet sexually mature and two calves have been observed in the Bering Sea (Carretta et al. 1994; Wade et al. 2006b). There is no evidence of reproductive success (i.e., young reared to independence) in the eastern North Pacific. No data are available for the western North Pacific.

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

Anthropogenic threats. Whaling for North Pacific right whales was discontinued in 1966 with the IWC whaling moratorium. North Pacific right whales remain at considerable risk of extinction, including: (1) life history characteristics such as slow growth rate, long calving intervals, and longevity; (2) distorted age structure of the population and reduced reproductive success; (3) strong depensatory or Allee effects; (4) habitat specificity or site fidelity; and (5) habitat sensitivity. The proximity of the other known right whale habitats to shipping lanes (e.g. Unimak Pass) suggests that collisions with vessels may also represent a threat to North Pacific right whales (Elvin and Hogart 2008).

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

Critical habitat. In July 2006, NMFS designated two areas as critical habitat for right whales in the North Pacific (71 FR 38277). The areas encompass about 36,750 square miles of marine habitat, which include feeding areas within the Gulf of Alaska and the Bering Sea that support the species. The primary constituent element to this critical habitat is the presence of large copepods and oceanographic factors that concentrate these prey of North Pacific right whales. At present, this PCE has not been significantly degraded due to human activity. Significant concern has been voiced regarding the impact that oceanic contamination of pollutants may have on the food chain and consequent bioaccumulation of toxins by marine predators. Changes due to global warming have also been raised as a concern that could affect the distribution or abundance of copepod prey for several marine mammals, including right whales.

Sei whale

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

Population designations. The population structure of sei whales is unknown and populations herein assume (based upon migratory patterns) population structuring is discrete by ocean basin (north and south), except for sei whales in the Southern Ocean, which may form a ubiquitous population or several discrete ones.

North Atlantic. In the western North Atlantic, a major portion of the sei whale population occurs in northern waters, potentially including the Scotian Shelf, along Labrador and Nova Scotia, south into the U.S. EEZ, including the Gulf of Maine and Georges Bank (Mitchell and Chapman 1977; Waring et al. 2004). These whales summer in northern areas before migrating south to waters along Florida, in the Gulf of Mexico, and the northern Caribbean Sea (Gambell 1985b; Mead 1977). Sei whales may range as far south as North Carolina. In the U.S. EEZ, the greatest abundance occurs during spring, with most sightings on the eastern edge of Georges Bank, in the Northeast Channel, and along the southwestern edge of Georges Bank in Hydrographer Canyon (CETAP 1982). In 1999, 2000, and 2001, NMFS aerial surveys found sei whales concentrated along the northern edge of Georges Bank during spring (Waring et al.

2004). Surveys in 2001 found sei whales south of Nantucket along the continental shelf edge (Waring et al. 2004). During years of greater prey abundance (e.g., copepods), sei whales are found in more inshore waters, such as the Great South Channel (in 1987 and 1989), Stellwagen Bank (in 1986), and the Gulf of Maine (Payne et al. 1990a; Schilling et al. 1992). In the eastern Atlantic, sei whales occur in the Norwegian Sea, occasionally occurring as far north as Spitsbergen Island, and migrate south to Spain, Portugal, and northwest Africa (Gambell 1985b; Jonsgård and Darling 1977; Olsen et al. 2009).

North Pacific. Some mark-recapture, catch distribution, and morphological research indicate more than one population may exist – one between 155°-175° W, and another east of 155° W (Masaki 1976; Masaki 1977). Sei whales have been reported primarily south of the Aleutian Islands, in Shelikof Strait and waters surrounding Kodiak Island, in the Gulf of Alaska, and inside waters of southeast Alaska and south to California to the east and Japan and Korea to the west (Leatherwood et al. 1982; Nasu 1974). Sei whales have been occasionally reported from the Bering Sea and in low numbers on the central Bering Sea shelf (Hill and DeMaster 1998b). Whaling data suggest that sei whales do not venture north of about 55°N (Gregr et al. 2000). Masaki (1977) reported sei whales concentrating in the northern and western Bering Sea from July-September, although other researchers question these observations because no other surveys have reported sei whales in the northern and western Bering Sea. Horwood (1987) evaluated Japanese sighting data and concluded that sei whales rarely occur in the Bering Sea. Horwood (1987) reported that 75-85% of the North Pacific population resides east of 180°. During winter, sei whales are found from 20°-23° N (Gambell 1985b; Masaki 1977).

Southern Hemisphere. Sei whales occur throughout the Southern Ocean during the austral summer, generally between 40°-50° S (Gambell 1985b). During the austral winter, sei whales occur off Brazil and the western and eastern coasts of southern Africa and Australia, although all of the 20 sightings off Argentina occurred in August or September (Iniguez et al. 2010). However, sei whales generally do not occur north of 30º S in the Southern Hemisphere (Reeves et al. 1999). However, confirmed sighting records exist for Papua New Guinea and New Caledonia, with unconfirmed sightings in the Cook Islands (Programme) 2007).

There is little information on the population structure of sei whales in the Antarctic; some degree of isolation appears to exist, although sei whale movements are dynamic and individuals move between stock designation areas (Donovan 1991; IWC 1980a).

Feeding. Sei whales are primarily planktivorous, feeding mainly on euphausiids and copepods, although they are also known to consume fish (Waring et al. 2006). In the Northern Hemisphere, sei whales consume small schooling fish such as anchovies, sardines, and mackerel when locally abundant (Konishi et al. 2009; Mizroch et al. 1984; Rice 1977). Sei whales in the North Pacific feed on euphausiids and copepods, which make up about 95% of their diets (Calkins 1986). The dominant food for sei whales off California during June-August is northern anchovy, while in September-October whales feed primarily on krill (Rice 1977). The balance of their diet consists of squid and schooling fish, including smelt, sand lance, Arctic cod, rockfish, pollack, capelin, and Atka mackerel (Nemoto and Kawamura 1977). In the Southern Ocean, analysis of stomach contents indicates sei whales consume *Calanus* spp. and small-sized euphasiids with prey composition showing latitudinal trends (Kawamura 1974). Sei whales in the Southern Hemisphere may reduce direct interspecific competition with blue and fin whales by consuming a wider variety of prey and by arriving later to feeding grounds (Kirkwood 1992). Rice (1977) suggested that the diverse diet of sei whales may allow them greater opportunity to take advantage of variable prey resources, but may also increase their potential for competition with commercial fisheries. In the North Pacific, sei whales appear to prefer feeding along the cold eastern currents (Perry et al. 1999). Sei whales have the flexibility to skim or engulf prey (Brodie and Vikingsson 2009).

Reproduction. Very little is known regarding sei whale reproduction. However, of 32 adult female sei whales harvested by Japanese whalers, 28 were found to be pregnant while one was pregnant and lactating during May-July 2009 cruises in the western North Pacific (Tamura et al. 2009).

Vocalization and hearing. Data on sei whale vocal behavior is limited, but includes records off the Antarctic Peninsula of broadband sounds in the 100-600 Hz range with 1.5 s duration and tonal and upsweep calls in the 200-600 Hz range of 1-3 s durations (McDonald et al. 2005). Differences may exist in vocalizations between ocean basins (Rankin and Barlow 2007). Vocalizations from the North Atlantic consisted of paired sequences (0.5-0.8 s, separated by 0.4-1.0 s) of 10-20 short (4 ms) FM sweeps between 1.5-3.5 kHz (Thomson and Richardson 1995).

Status and trends. The sei whale was originally listed as endangered in 1970 (35 FR 18319), and this status remained since the inception of the ESA in 1973. Table 6 provides estimates of historic and current abundance for ocean regions.

Table 6. Summary of past and present sei whale abundance.

Region	Population, stock, or study area	Pre-exploitation estimate	95% C.I.	Current estimate	95% C.I.	Source
Global	--	>105,000	--	25,000	--	(Braham 1991)
North Atlantic	Basinwide	--	--	>4000	--	(Braham 1991)
	NMFS - Nova Scotia stock	--	--	386	--	(Angliss and Outlaw 2009)
	IWC - Iceland-Denmark stock	--	--	1,290	0-2,815*	(Cattanach et al. 1993)
	IWC - Iceland-Denmark stock	--	--	1,590	343-2,837*	(Cattanach et al. 1993)
North Pacific	Basinwide	42,000	--	7,260-12,620*	--	(Tillman 1977); *circa 1974
	NMFS - eastern North Pacific stock	--	--	46	CV=0.61	(Carretta et al. 2008)
	NMFS - Hawaii stock	--	--	77	0-237*	(Carretta et al. 2008)
	Western North Pacific	--	--	68,000	31,000-149,000	Hakamada et al. (2004)
Southern Hemisphere	Basinwide	63,100	--	--	--	(Mizroch et al. 1984)
	Basinwide	65,000	--	--	--	(Braham 1991)

South of 60ºS	--	--	626	553-699	(IWC 1996)
South of 30ºS	--	--	9,718	--	(IWC 1996)

*Note: Confidence Intervals (C.I.) not provided by the authors were calculated from Coefficients of Variation (C.V.) where available, using the computation from Gotelli and Ellison (2004).

North Atlantic. No information on sei whale abundance exists prior to commercial whaling (Perry et al. 1999). Between 1966 and 1972, whalers from land stations on the east coast of Nova Scotia engaged in extensive hunts of sei whales on the Nova Scotia shelf, killing about 825 sei whales (Mitchell and Chapman 1977). In 1974, the North Atlantic stock was estimated to number about 2,078 individuals, including 965 whales in the Labrador Sea group and 870 whales in the Nova Scotia group (Mitchell and Chapman 1977). In the northwest Atlantic, Mitchell and Chapman (1977) estimated the Nova Scotia stock to contain between 1,393-2,248 whales; and an aerial survey program conducted from 1978 to 1982 on the continental shelf and edge between Cape Hatteras, North Carolina, and Nova Scotia generated an estimate of 280 sei whales (CETAP 1982). These two estimates are more than 20 years out of date and likely do not reflect the current true abundance; in addition, the Cetacean and Turtle Assessment Program estimate has a high degree of uncertainty and is considered statistically unreliable (Perry et al. 1999; Waring et al. 2004; Waring et al. 1999). The total number of sei whales in the U.S. Atlantic EEZ remains unknown (Waring et al. 2006). Rice (1977) estimated total annual mortality for adult females as 0.088 and adult males as 0.103.

North Pacific. Ohsumi and Fukuda (1975) estimated that sei whales in the North Pacific numbered about 49,000 whales in 1963, had been reduced to 37,000-38,000 whales by 1967, and reduced again to 20,600-23,700 whales by 1973. From 1910-1975, approximately 74,215 sei whales were caught in the entire North Pacific Ocean (Horwood 1987; Perry et al. 1999). From the early 1900s, Japanese whaling operations consisted of a large proportion of sei whales, killing 300-600 sei whales per year from 1911-1955. The sei whale catch peaked in 1959, when 1,340 sei whales died. In 1971, after a decade of high sei whale catch numbers, sei whales were scarce in Japanese waters. Japanese and Soviet catches of sei whales in the North Pacific and Bering Sea increased from 260 whales in 1962 to over 4,500 in 1968-1969, after which the sei whale population declined rapidly (Mizroch et al. 1984). When commercial whaling for sei whales ended in 1974, the population in the North Pacific had been reduced to 7,260-12,620 animals (Tillman 1977). There have been no direct estimates of sei whale populations for the eastern Pacific Ocean (or the entire Pacific). Between 1991-2001, during aerial surveys, there were two confirmed sightings of sei whales along the U.S. Pacific coast. The minimum estimate of individuals along the U.S. west coast between 1996-2001 was 35 (Carretta et al. 2006).

Natural threats. The foraging areas of right and sei whales in the western North Atlantic Ocean overlap and both whales feed preferentially on copepods (Mitchell 1975).

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

Anthropogenic threats. Human activities known to threaten sei whales include whaling, commercial fishing, and maritime vessel traffic. Historically, whaling represented the greatest threat to every population of sei whales and was ultimately responsible for listing sei whales as an endangered species. Sei whales are thought to not be widely hunted, although harvest for scientific whaling or illegal harvesting occurs in some areas. Japan maintains an active whaling fleet, killing up to 101 sei whales annually (IWC 2008). In 2009, 100 sei whales were killed during western North Pacific surveys (Bando et al. 2010).

Sei whales occasionally die in collisions with vessels. Of three sei whales that stranded along the U.S. Atlantic coast between 1975-1996, two showed evidence of collisions (Laist et al. 2001). Between 1999 and 2005, there were three reports of sei whales being struck by vessels along the U.S. Atlantic coast and Canada's Maritime Provinces (Cole et al. 2005b; Nelson et al. 2007b). Two of these ship strikes reportedly resulted in death. One sei whale was killed in a collision with a vessel off the coast of Washington in 2003 (Waring et al. 2008). New rules for seasonal (June through December) slowing of vessel traffic in the Bay of Fundy to 10 knots and changing shipping lanes by less than one nautical mile to avoid the greatest concentrations of right whales are predicted to reduce sei whale ship strike mortality by 17%.

Sei whales accumulate DDT, DDE, and PCBs (Borrell 1993; Borrell and Aguilar 1987; Henry and Best 1983). Males carry larger burdens than females, as gestation and lactation transfer these toxins from mother to offspring. In the northwestern Pacific, PCB levels in baleen whales appear to have declined through the 1990s before plateauing in the 2000s (Yasunaga and Fujise 2009b).

Critical habitat. The NMFS has not designated critical habitat for sei whales.

Sperm whale

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

Stock designations. There is no clear understanding of the global population structure of sperm whales (Dufault et al. 1999). Recent ocean-wide genetic studies indicate low, but statistically significant, genetic diversity and no clear geographic structure, but strong differentiation between social groups (Lyrholm and Gyllensten 1998; Lyrholm et al. 1996; Lyrholm et al. 1999). The IWC currently recognizes four sperm whale stocks: North Atlantic, North Pacific, northern Indian Ocean, and Southern Hemisphere (Dufault et al. 1999; Reeves and Whitehead 1997). The NMFS recognizes six stocks under the MMPA- three in the Atlantic/Gulf of Mexico and three in the Pacific (Alaska, California-Oregon-Washington, and Hawaii; (Perry et al. 1999; Waring et al. 2004). Genetic studies indicate that movements of both sexes through expanses of ocean basins are common, and that males, but not females, often breed in different ocean basins than the ones in which they were born (Whitehead 2003a). Sperm whale populations appear to be structured socially, at the level of the clan, rather than geographically (Whitehead 2003a; Whitehead et al. 2008).

North Atlantic. In the western North Atlantic, sperm whales range from Greenland south into the Gulf of Mexico and the Caribbean, where they are common, especially in deep basins north of the continental shelf (Romero et al. 2001; Wardle et al. 2001). The northern distributional limit of female/immature pods is probably around Georges Bank or the Nova

Scotian shelf (Whitehead et al. 1991). Seasonal aerial surveys confirm that sperm whales are present in the northern Gulf of Mexico in all seasons (Hansen et al. 1996; Mullin et al. 1994). Sperm whales distribution follows a distinct seasonal cycle, concentrating east-northeast of Cape Hatteras in winter and shifting northward in spring when whales occur 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, mature male sperm whales have been recorded as far north as Spitsbergen (Øien 1990). Recent observations of sperm whales and stranding events involving sperm whales from the eastern North Atlantic suggest that solitary and paired mature males predominantly occur in waters off Iceland, the Faroe Islands, and the Norwegian Sea (Christensen et al. 1992a; Christensen et al. 1992b; Gunnlaugsson and Sigurjónsson 1990; Øien 1990).

North Pacific. Sperm whales are found throughout the North Pacific and are distributed broadly in tropical and temperate waters to the Bering Sea as far north as Cape Navarin in summer, and occur south of 50^{o} N in winter (Gosho et al. 1984; Miyashita et al. 1995 as cited in Carretta et al. 2005; Rice 1974; Whitehead 2003b), with males exhibiting a somewhat broader latitudinal range. Sperm whales are found year-round in Californian and Hawaiian waters (Barlow 1995; Dohl et al. 1983; Forney et al. 1995; Lee 1993; Mobley Jr . et al. 2000; Rice 1960; Shallenberger 1981), but they reach peak abundance from April-mid-June and from the end of August-mid-November (Rice 1974). They are seen in every season except winter (December-February) in Washington and Oregon (Green et al. 1992). Summer/fall surveys in the eastern tropical Pacific (Wade and Gerrodette 1993) show that although sperm whales are widely distributed in the tropics, their relative abundance tapers off markedly towards the middle of the tropical Pacific and northward towards the tip of Baja California (Carretta et al. 2006). Sperm whales occupying the California Current region are genetically distinct from those in the eastern tropical Pacific and Hawaiian waters (Mesnick et al. 2011). The discreteness of the later two areas remains uncertain (Mesnick et al. 2011).

Mediterranean. Sperm whales occur from the Alboran Sea to the Levant Basin, primarily over steep slope and deep offshore waters. Sperm whales are rarely sighted in the Sicilian Channel, and are vagrants to the northern Adriatic and Aegean Seas (Notarbartolo di Sciara and Demma 1997). In 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.

Southern Hemisphere. All sperm whales of the Southern Hemisphere are treated as a single stock with nine divisions, although this designation has little biological basis and is more in line with whaling records (Donovan 1991). Sperm whales that occur off the Galapagos Islands, mainland Ecuador, and northern Peru may be distinct from other sperm whales in the Southern Hemisphere (Dufault and Whitehead 1995; Rice 1977; Wade and Gerrodette 1993). Gaskin (1973) found females to be absent in waters south of 50^{o} and decrease in proportion to males south of 46-47^{o}.

Movement. Movement patterns of Pacific female and immature male groups appear to follow prey distribution and, although not random, movements are difficult to anticipate and are likely associated with feeding success, perception of the environment, and memory of optimal foraging areas (Whitehead et al. 2008). No sperm whale in the Pacific has been known to travel to points over 5,000 km apart and only rarely have been known to move over 4,000 km within a time frame of several years. This means that although sperm whales do not appear to cross from eastern to western sides of the Pacific (or vice-versa), significant mixing occurs that can maintain genetic exchange. Movements of several hundred miles are common, (i.e. between the Galapagos Islands and the Pacific coastal Americas). Movements appear to be group or clan specific, with some groups traveling straighter courses than others over the course of several days. General transit speed averages about 4 km/h. Sperm whales in the Caribbean region appear to be much more restricted in their movements, with individuals repeatedly sighted within less than 160 km of previous sightings.

Gaskin (1973) proposed a northward population shift of sperm whales off New Zealand in the austral autumn based on reduction of available food species and probable temperature tolerances of calves.

Habitat. Sperm whales have a strong preference for waters deeper than 1,000 m (Reeves and Whitehead 1997; Watkins 1977), although Berzin (1971) reported that they are restricted to waters deeper than 300 m. While deep water is their typical habitat, sperm whales are rarely found in waters less than 300 m in depth (Clarke 1956; Rice 1989a). Sperm whales have been observed near Long Island, New York, in water between 40-55 m deep (Scott and Sadove 1997). When they are found relatively close to shore, sperm whales are usually associated with sharp increases in topography where upwelling occurs and biological production is high, implying the presence of a good food supply (Clarke 1956). Such areas include oceanic islands and along the outer continental shelf.

Sperm whales are frequently found in locations of high productivity due to upwelling or steep underwater topography, such as continental slopes, seamounts, or canyon features (Jaquet and Whitehead 1996; Jaquet et al. 1996). Cold-core eddy features are also attractive to sperm whales in the Gulf of Mexico, likely because of the large numbers of squid that are drawn to the high concentrations of plankton associated with these features (Biggs et al. 2000; Davis et al. 2000a; Davis et al. 2000b; Davis et al. 2000c; Davis et al. 2002; Wormuth et al. 2000). Surface waters with sharp horizontal thermal gradients, such as along the Gulf Stream in the Atlantic, may also be temporary feeding areas for sperm whales (Griffin 1999; Jaquet et al. 1996; Waring et al. 1993). Sperm whale over George's Bank were associated with surface temperatures of 23.2-24.9°C (Waring et al. 2003).

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

Sperm whale age distribution is unknown, but sperm whales are believed to live at least 60 years

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

Stable, long-term associations among females form the core of sperm whale societies (Christal et al. 1998). Up to about a dozen females usually live in such groups, accompanied by their female and young male offspring. Young individuals are subject to alloparental care by members of either sex and may be suckled by non-maternal individuals (Gero et al. 2009). Group sizes may be smaller overall in the Caribbean Sea (6-12 individuals) versus the Pacific (25-30 individuals)(Jaquet and Gendron 2009). Groups may be stable for long periods, such as for 80 days in the Gulf of California (Jaquet and Gendron 2009). Males start leaving these family groups at about 6 years of age, after which they live in “bachelor schools,” but this may occur more than a decade later (Pinela et al. 2009). The cohesion among males within a bachelor school declines with age. During their breeding prime and old age, male sperm whales are essentially solitary (Christal and Whitehead 1997).

Diving. Sperm whales are probably the deepest and longest diving mammalian species, with dives to 3 km and durations in excess of 2 hours (Clarke 1976; Watkins et al. 1993; Watkins et al. 1985). However, dives are generally shorter (25- 45 min) and shallower (400-1,000 m). Dives are separated by 8-11 min rests at the surface (Gordon 1987; Jochens et al. 2006; Papastavrou et al. 1989; Watwood et al. 2006; Würsig et al. 2000). Sperm whales typically travel ~3 km horizontally and 0.5 km vertically during a foraging dive (Whitehead 2003a). Differences in night and day diving patterns are not known for this species, but, like most diving air-breathers for which there are data (rorquals, fur seals, and chinstrap penguins), sperm whales probably make relatively shallow dives at night when prey are closer to the surface.

Feeding. Sperm whales appear to feed regularly throughout the year (NMFS 2006c). It is estimated they consume about 3-3.5% of their body weight daily (Lockyer 1981). They seem to forage mainly on or near the bottom, often ingesting stones, sand, sponges, and other non-food items (Rice 1989a). A large proportion of a sperm whale’s diet consists of low-fat, ammoniacal, or luminescent squids (Clarke 1996; Clarke 1980b; Martin and Clarke 1986). While sperm whales feed primarily on large and medium-sized squids, the list of documented food items is fairly long and diverse. Prey items include other cephalopods, such as octopi, and medium- and large-sized demersal fishes, such as rays, sharks, and many teleosts (Angliss and Lodge 2004; Berzin 1972; Clarke 1977; Clarke 1980a; Rice 1989a). The diet of large males in some areas, especially in high northern latitudes, is dominated by fish (Rice 1989a). In some areas of the North Atlantic, however, males prey heavily on the oil-rich squid *Gonatus fabricii*, a species also frequently eaten by northern bottlenose whales (Clarke 1997).

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

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

Status and trends. Sperm whales were originally listed as endangered in 1970 (35 FR 18319), and this status remained with the inception of the ESA in 1973. Although population structure of sperm whales is unknown, several studies and estimates of abundance are available. Table 7 contains historic and current estimates of sperm whales by region. Sperm whale populations probably are undergoing the dynamics of small population sizes, which is a threat in and of itself. In particular, the loss of sperm whales to directed Soviet whaling likely inhibits recovery due to the loss of adult females and their calves, leaving sizeable gaps in demographic and age structuring (Whitehead 2003a).

North Atlantic. 190,000 sperm whales were estimated to have been in the entire North Atlantic, but CPUE data from which this estimate is derived are unreliable according to the IWC (Perry et al. 1999). The total number of sperm whales in the western North Atlantic is unknown (Waring et al. 2008). The best available current abundance estimate for western North Atlantic sperm whales is 4,804 based on 2004 data. The best available estimate for Northern Gulf of Mexico sperm whales is 1,665, based on 2003-2004 data, which are insufficient to determine population trends (Waring et al. 2008). Sperm whales were widely harvested, from the northeastern Caribbean (Romero et al. 2001) and the Gulf of Mexico where a sperm whale fishery operated during the late 1700s to the early 1900s (NMFS 2006c; Townsend 1935b).

Table 7. Summary of past and present sperm whale abundance.

Region	Population, stock, or study area	Pre-exploitation estimate	95% C.I.	Current estimate	95% C.I.	Source
Global	--	--	--	900,000	--	(Würsig et al. 2000)
	--	1,100,000	670,000-1,512,000	360,000	105,984-614,016*	(Whitehead 2002)
North Atlantic	Basinwide	224,800	--	22,000	--	(Gosho et al. 1984; Würsig et al. 2000)

Region	Population, stock, or study area	Pre-exploitation estimate	95% C.I.	Current estimate	95% C.I.	Source
	Northeast Atlantic, Faroes-Iceland, and U.S. East coast	--	--	13,190	--	(Whitehead 2002)
	NMFS-North Atlantic stock (Western North Atlantic)	--	--	4,804	1,226-8,382*	(NMFS 2008)
	Eastern North Atlantic-Iceland	--	--	1,234	823-1,645*	(Gunnlaugsson and Sigurjónsson 1990)
	Eastern North Atlantic-Faroe Islands	--	--	308	79-537*	(Gunnlaugsson and Sigurjónsson 1990)
	Eastern North Atlantic-Norwegian Sea	--	--	5,231	2,053-8,409*	(Christensen et al. 1992b)
	Eastern North Atlantic-Northern Norway to Spitsbergen	--	--	2,548	1,200-3,896*	(Øien 1990)
Gulf of Mexico	NMFS-Gulf of Mexico stock	--	--	1,665	CV=0.2	(Waring et al. 2009)
	Northern Gulf of Mexico – off the Mississippi River Delta between 86º and 91º W	--	--	398	253-607	(Jochens et al. 2006)
	North-central and northwestern Gulf of Mexico	--	--	87	52-146	(Mullin et al. 2004)
North Pacific	Basinwide	620,400	--	472,100930,000	--	(Gosho et al. 1984)
		--	--	930,000	--	(Rice 1989a)
	Eastern tropical Pacific	--	--	26,053	13,797-38,309*	(Whitehead 2003a)
	Off Costa Rica	--	--	1,360	823-2,248*	(Gerrodette and Palacios 1996)

Region	Population, stock, or study area	Pre-exploitation estimate	95% C.I.	Current estimate	95% C.I.	Source
	Off Central America north of Costa Rica	--	--	333	125-890*	(Gerrodette and Palacios 1996)
	Eastern temperate North Pacific	--	--	26,300	0-68,054	(Barlow and Taylor 2005)
		--	--	32,100	9,450-54,750*	(Barlow and Taylor 2005)
	NMFS-California/ Oregon/Washington stock	--	--	2,853	CV=0.25*	(Carretta et al. 2008)
	NMFS-Hawaii stock	--	--	7,082	2,918-11,246*	(Carretta et al. 2008)
	Western North Pacific	--	--	102,000	75,000-148,000	Kato and Miyashita (2000)
Southern Hemisphere	Basinwide	547,600	--	299,400	--	(Gosho et al. 1984; IWC 1988; Perry et al. 1999)
	South of 60^{o}S	--	--	14,000	8,786-19,214	(Butterworth et al. 1995) as cited in (Perry et al. 1999)
	South of 30^{o}S	--	--	128,000	17,613-238,687	(Butterworth et al. 1995) as cited in (Perry et al. 1999)

*Note: Confidence Intervals (C.I.) not provided by the authors were calculated from Coefficients of Variation (C.V.) where available, using the computation from Gotelli and Ellison (2004).

North Pacific. There are approximately 76,803 sperm whales in the eastern tropical Pacific, eastern North Pacific, Hawaii, and western North Pacific (Whitehead 2002). Minimum estimates in the eastern North Pacific are 1,719 individuals and 5,531 in the Hawaiian Islands (Carretta et al. 2007b). The tropical Pacific is home to approximately 26,053 sperm whales and the western North Pacific has approximately 29,674 (Whitehead 2002). There was a dramatic decline in the number of females around the Galapagos Islands during 1985-1999 versus 1978-1992 levels, likely due to migration to nearshore waters of South and Central America (Whitehead 2003a).

Hill and DeMaster (1999) concluded that about 258,000 sperm whales were harvested in the North Pacific between 1947-1987. Although the IWC protected sperm whales from commercial harvest in 1981, Japanese whalers continued to hunt sperm whales in the North Pacific until 1988 (Reeves and Whitehead 1997). In 2000, the Japanese Whaling Association announced plans to kill 10 sperm whales in the Pacific Ocean for research. Although consequences of these deaths are unclear, the paucity of population data, uncertainly regarding recovery from whaling, and re-establishment of active programs for whale harvesting pose risks for the recovery and survival of this species. Whalers hunted for subsistence purposes from

Lamalera, Indonesia, where a traditional whaling industry reportedly kills up to 56 sperm whales per year.

Southern Hemisphere. Whaling in the Southern Hemisphere averaged roughly 20,000 whales between 1956-1976 (Perry et al. 1999). Population size appears to be stable (Whitehead 2003a). Whitehead (2002b) estimated 12,069 sperm whales south of 60° S.

Natural threats. Sperm whales are known to be occasionally predated upon by killer whales (Jefferson and Baird 1991; Pitman et al. 2001) and large sharks (Best et al. 1984) and harassed by pilot whales (Arnbom et al. 1987; Palacios and Mate 1996; Rice 1989b; Weller et al. 1996; Whitehead 1995). Strandings are also relatively common events, with one to dozens of individuals generally beaching themselves and dying during any single event. Although several hypotheses, such as navigation errors, illness, and anthropogenic stressors, have been proposed (Goold et al. 2002; Wright 2005), direct widespread causes remain unclear. Calcivirus and papillomavirus are known pathogens of this species (Lambertsen et al. 1987; Smith and Latham 1978).

Anthropogenic threats. Sperm whales historically faced severe depletion from commercial whaling operations. From 1800 to 1900, the IWC estimated that nearly 250,000 sperm whales were killed by whalers, with another 700,000 from 1910 to 1982 (IWC Statistics 1959-1983). Other estimates have included 436,000 individuals killed between 1800-1987 (Carretta et al. 2005). All of these estimates are likely underestimates due to illegal and inaccurate killings by Soviet whaling fleets between 1947 and 1973. In the Southern Hemisphere, these whalers killed an estimated 100,000 whales that they did not report to the IWC (Yablokov et al. 1998), with smaller harvests in the Northern Hemisphere, primarily the North Pacific, that extirpated sperm whales from large areas (Yablokov and Zemsky 2000). Additionally, Soviet whalers disproportionately killed adult females in any reproductive condition (pregnant or lactating) as well as immature sperm whales of either gender.

Following a moratorium on whaling by the IWC, large-scale commercial whaling pressures on sperm whales ended. However, Japan maintains an active whaling fleet, killing up to 10 sperm whales annually (IWC 2008). In 2009, one sperm whale was killed during western North Pacific surveys (Bando et al. 2010).

Sperm whales are also known to have become entangled in commercial fishing gear and 17 individuals are known to have been struck by vessels (Jensen and Silber 2004). Between 1998 and 2005, observers identified 6 sperm whales injured or killed by fisheries off the U.S. west coast (NMFS, unpublished data). Whale-watching vessels are known to influence sperm whale behavior (Richter et al. 2006). Sperm whales are also killed incidentally by gill nets at a rate of roughly nine per year (data from 1991 to 1995) in U.S. Pacific waters (Barlow et al. 1997). Sperm whales interact with (remove fish from) longline fisheries in the Gulf of Alaska and entanglement has rarely been recorded (Hill and DeMaster 1999; Rice 1989a; Sigler et al. 2008).

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

Contaminants have been identified in sperm whales, but vary widely in concentration based upon life history and geographic location, with northern hemisphere individuals generally carrying higher burdens (Evans et al. 2004). Contaminants include dieldrin, chlordane, DDT, DDE, PCBs, HCB and HCHs in a variety of body tissues (Aguilar 1983; Evans et al. 2004), as well as several heavy metals (Law et al. 1996) (Yasunaga and Fujise 2009a). Unlike other marine mammals, females appear to bioaccumulate toxins at greater levels than males, which may be related to possible dietary differences between females who remain at relatively low latitudes compared to more migratory males (Aguilar 1983; Wise et al. 2009; Yasunaga and Fujise 2009a). Chromium levels from sperm whale skin samples worldwide have varied from undetectable to 122.6 μg Cr/g tissue, with the mean (8.8 μg Cr/g tissue) resembling levels found in human lung tissue with chromium-induced cancer (Wise et al. 2009). Older or larger individuals did not appear to accumulate chromium at higher levels.

Whale watching is a rapidly-growing activity in Japan, with over 45 businesses operating 185 vessels in 30 communities (Hoyt 1993; Hoyt 2001). Sperm whales are targeted in several locations (Hoyt 1993; Hoyt 2001; Mori 1999; Uchida 1997). Peak season is in winter through summer, depending upon location and target species (Hoyt 1993).
Critical habitat. The NMFS has not designated critical habitat for sperm whales.

Sea turtles

Green sea turtle

Distribution. Green sea turtles have a circumglobal distribution, occurring throughout tropical, subtropical waters, and, to a lesser extent, temperate waters.

Population designation. Populations are distinguished generally by ocean basin and more specifically by nesting location (Table 8).

Based upon genetic differences, two or three distinct regional clades may exist in the Pacific: western Pacific and South Pacific islands, eastern Pacific, and central Pacific, including the rookery at French Frigate Shoals, Hawaii (Dutton and Balazs In review; Dutton et al. 1996). In the eastern Pacific, green sea turtles forage from San Diego Bay, California to Mejillones, Chile. Individuals along the southern foraging area originate from Galapagos Islands nesting beaches, while those in the Gulf of California originate primarily from Michoacán. Green turtles foraging in San Diego Bay and along the Pacific coast of Baja California originate primarily from rookeries of the Islas Revillagigedos (Dutton 2003a).

Table 8. Locations and most recent abundance estimates of threatened green sea turtles as annual nesting females (AF), annual nests (AN), annual egg production (EP), and annual egg harvest (EH).

Location	Most recent abundance	Reference
Western Atlantic Ocean		

Location	Most recent abundance	Reference
Tortuguero, Costa Rica	17,402-37,290 AF	(Troëng and Rankin 2005)
Aves Island, Venezuela	335-443 AF	(Vera 2007)
Galibi Reserve, Suriname	1,803 AF	(Weijerman et al. 1998)
Isla Trindade, Brazil	1,500-2,000 AF	(Moreira and Bjorndal 2006)
Central Atlantic Ocean		
Ascension Island, UK	3,500 AF	(Broderick et al. 2006)
Eastern Atlantic Ocean		
Poilao Island, Guinea-Bissau	7,000-29,000 AN	(Catry et al. 2009)
Bioko Island, Equatorial Guinea	1,255-1,681 AN	(Tomas et al. 1999)
Mediterranean Sea		
Turkey	214-231 AF	(Broderick et al. 2002)
Cyprus	121-127 AF	(Broderick et al. 2002)
Israel / Palestine	1-3 AF	(Kuller 1999)
Syria	100 AN	(Rees et al. 2005)
Western Indian Ocean		
Eparces Islands	2,000-11,000 AF	(Le Gall et al. 1986)
Comoros Islands	5,000 AF	S. Ahamada, pers. comm. 2001
Seychelles Islands	3,535-4,755 AF	J. Mortimer, pers. comm. 2002
Kenya	200-300 AF	(Okemwa and Wamukota 2006)
Northern Indian Ocean		
Ras al Hadd, Oman	44,000 AN	S. Al-Saady, pers. comm. 2007
Sharma, Yemen	15 AF	(Saad 1999)
Karan Island, Saudi Arabia	408-559 AF	(Pilcher 2000)
Jana and Juraid Islands, Saudi Arabia	643 AN	(Pilcher 2000)
Hawkes Bay and Sandspit, Pakistan	600 AN	(Asrar 1999)
Gujarat, India	461 AN	(Sunderraj et al. 2006)
Sri Lanka	184 AF	(Kapurisinghe 2006)
Eastern Indian Ocean		
Thamihla Kyun, Myanmar	<250,000 EH	(Thorbjarnarson et al. 2000)
Pangumbahan, Indonesia	400,000 EH	(Schulz 1987)
Suka Made, Indonesia	395 AN	C. Limpus, pers. comm. 2002
Western Australia	3,000-30,000 AN	R. Prince, pers. comm. 2001
Southeast Asia		
Gulf of Thailand	250 AN	Charuchinda pers. comm. 2001
Vietnam	239 AF	(Hamann et al. 2006b)
Berau Islands, Indonesia	4,000-5,000 AF	(Schulz 1984)
Turtle Islands, Philippines	1.4 million EP	(Cruz 2002)
Sabah Turtle Islands, Malaysia	8,000 AN	(Chan 2006)
Sipadan, Malaysia	800 AN	(Chan 2006)
Sarawak, Malaysia	2,000 AN	(Liew 2002)
Enu Island (Aru Islands)	540 AF	Dethmers, in preparation
Terengganu, Malaysia	2,200 AN	(Chan 2006)
Western Pacific Ocean		
Heron Island, Australia	560 AF	(Limpus et al. 2002)
Raine Island, Australia	25,000 AF	(Limpus et al. 2003)
Guam	45 AF	(Cummings 2002)
Ogasawara Islands, Japan	500 AF	(Chaloupka et al. 2007)
Central and Eastern Pacific Ocean		
French Frigate Shoals, Hawaii	400 AF	(Balazs and Chaloupka 2006)
Michoacán, Mexico	1,395 AF	C. Delgado, pers. comm. 2006
Central American Coast	184-344 AN	(López and Arauz 2003)
Galapagos Islands, Ecuador	1,650 AF	(Zárate et al. 2006)

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

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

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

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

Migration and movement. Green sea turtles are highly mobile and undertake complex movements through geographically disparate habitats during their lifetimes (Musick and Limpus 1997; Plotkin 2003). The periodic migration between nesting sites and foraging areas by adults is a prominent feature of their life history. After departing as hatchlings and residing in a variety of marine habitats for 40 or more years (Limpus and Chaloupka 1997), green sea turtles make their way back to the same beach from which they hatched (Carr et al. 1978; Meylan et al. 1990). Green sea turtles spend the majority of their lives in coastal foraging grounds. These areas include both open coastline and protected bays and lagoons. While in these areas, green sea turtles rely on marine algae and seagrass as their primary dietary constituents, although some populations also forage heavily on invertebrates. There is some evidence that individuals move from shallow seagrass beds during the day to deeper areas at night (Hazel 2009). However, avoidance of areas of greater than 10 m when moderate depths of 5-10 m with sea grass beds has been found, with speed and displacement from capture locations being similar at night as during the daytime (Senko et al. 2010a).

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

In the western Pacific, green sea turtles forage along the coast of Japan as well as in the East China Sea and off Taiwan (Suganuma 1989; Tachikawa et al. 1994). Sightings along Japan are most frequent along Honshu Island, north to 35º N, as well as along the Ogasawara Islands during nesting season (May through early September, peaking in June and July); individuals seem to move between these two locations (Chan et al. 2007a; Eckert 1993b; Suganuma 1989; Suganuma et al. 1996; Tachikawa et al. 1994; Uchida and Nishiwaki 1995). The Ogasawara Islands are one of the most important nesting areas for green sea turtles in the western Pacific. Females return to nest here every 2-9 years, with a mode of four years between nesting events (Tachikawa et al. 1994). For each nesting year, females lay an average of four clutches, each containing slightly more than 100 eggs (Suganuma et al. 1996). Nesting in Japan also occurs along the Ryukyu Islands in May through August, which represents the northern extent of green sea turtle nesting in the western Pacific (Abe et al. 1998; Chan et al. 2007a; Kikukawa et al. 1996; Uchida 1994). Migrations between foraging and nesting habitats appear to occur through coastal waters (Tachikawa et al. 1994; Uchida 1994; Uchida and Nishiwaki 1995). Individuals foraging along the Japanese coast may also nest along China and Taiwan (Cheng 2000; Song et al. 2002). Green sea turtles have not been bycaught by longliners in the Shatsky Rise region (Yokota et al. 2006).

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

Diving. Based on the behavior of post-hatchlings and juvenile green turtles raised in captivity, we presume that those in pelagic habitats live and feed at or near the ocean surface, and that their dives do not normally exceed several meters in depth (Hazel et al. 2009; NMFS and USFWS 1998a). Recent data from Australia indicate green sea turtles rarely dive deep, staying in upper 8 m of the water column (Hazel et al. 2009). Here, daytime dives were shorter and shallower than were nighttime dives. Also, time spent resting and dive duration increased significantly with decreases in seasonal water temperatures. The maximum recorded dive depth for an adult green turtle was just over 106 m (Berkson 1967), while subadults routinely dive to 20 m for 9-23 min, with a maximum recorded dive of over 1 h (Brill et al. 1995; I-Jiunn 2009). Green sea turtles along Taiwan may rest during long, shallow dives (I-Jiunn 2009). Dives by females may be shorter in the period leading up to nesting (I-Jiunn 2009).

Vocalization and hearing. Although very limited information is available regarding green turtle hearing, it is one of the few sea turtle species that have been studied. Based upon auditory brainstem responses of three green sea turtles in air, these individuals had maximum sensitivity to sound in the 300-400 Hz range. A similar study by Bartol and Ketten (2006) found a range of 100-500 Hz as being sensitive to sound, with maximum sensitivity from 200-400 Hz. These same authors also reported a pair of juvenile green turtles to be generally sensitive between 100-800 Hz and most sensitive between 600-700 Hz. Outside of this limited range, green turtles are much less sensitive to sound (Ridgway et al. 1969). This is similar to estimates for loggerhead sea turtles, which have most sensitive hearing between 250-1,000 Hz, with rapid decline above 1,000 Hz (Moein Bartol et al. 1999).

Status and trends. Federal listing of the green sea turtle occurred on July 28, 1978, with all populations listed as threatened except for the Florida and Pacific coast of Mexico breeding populations, which are endangered (43 FR 32800). The International Union for Conservation of Nature (IUCN) has classified the green turtle as "endangered."

No trend data are available for almost half of the important nesting sites, where numbers are based on recent trends and do not span a full green sea turtle generation, and impacts occurring over four decades ago that caused a change in juvenile recruitment rates may have yet to be manifested as a change in nesting abundance. The numbers also only reflect one segment of the population (nesting females), who are the only segment of the population for which reasonably good data are available and are cautiously used as one measure of the possible trend of populations.

Table 9 summarizes nesting abundance for 46 nesting sites worldwide. These include both large and small rookeries believed to be representative of the overall trends for their respective regions. Based on the mean annual reproductive effort, 108,761-150,521 females nest each year among the 46 sites. Overall, of the 26 sites for which data enable an assessment of current trends, 12 nesting populations are increasing, 10 are stable, and four are decreasing. Long-term continuous datasets of 20 years are available for 11 sites, all of which are either increasing or stable. Despite the apparent global increase in numbers, the positive overall trend should be viewed cautiously because trend data are available for just over half of all sites examined and very few data sets span a full green sea turtle generation (Seminoff 2004a).

Pacific Ocean. Green turtles are thought to be declining throughout the Pacific Ocean, with the exception of Hawaii, from a combination of overexploitation and habitat loss (Eckert 1993b; Seminoff et al. 2002a). In the western Pacific, the only major (>2,000 nesting females) populations of green turtles occur in Australia and Malaysia, with smaller colonies throughout the area. Indonesian nesting is widely distributed, but has experienced large declines over the past 50 years. Hawaii green turtles are genetically distinct and geographically isolated, and the population appears to be increasing in size despite the prevalence of fibropapillomatosis and spirochidiasis (Aguirre et al. 1998).

The East Island nesting beach in Hawaii is showing a 5.7% annual growth rate over >25 years (Chaloupka et al. 2008a). In the Eastern Pacific, mitochondrial DNA analysis has indicated three key nesting populations: Michoacán, Mexico; Galapagos Islands, Ecuador; and Islas Revillagigedos, Mexico (Dutton 2003b). The number of nesting females per year exceeds 1,000 females at each site (NMFS and USFWS 2007a). However, historically, >20,000 females per year are believed to have nested in Michoacán alone (Clifton et al. 1982; NMFS and USFWS 2007a). Thus, the current number of nesting females is still far below historical levels. Datasets over 25 years in Chichi-jima, Japan; Heron Island, Australia; and Raine Island, Australia, show increases in abundance (Chaloupka et al. 2008a).

Atlantic Ocean. Primary sites for green sea turtle nesting in the Atlantic/Caribbean include: (1) Yucatán Peninsula, Mexico; (2) Tortuguero, Costa Rica; (3) Aves Island, Venezuela; (4) Galibi Reserve, Suriname; (5) Isla Trindade, Brazil; (6) Ascension Island, United Kingdom; (7) Bioko Island, Equatorial Guinea; and (8) Bijagos Archipelago, Guinea-Bissau (NMFS and USFWS 2007a). Nesting at all of these sites was considered to be stable or increasing with the exception of Bioko Island and the Bijagos Archipelago where the lack of sufficient data precludes a meaningful trend assessment for either site (NMFS and USFWS 2007a). Seminoff (2004b) reviewed green sea turtle nesting data for eight sites in the western, eastern, and central Atlantic. Seminoff (2004b) concluded that all sites in the central and western Atlantic showed increased nesting, with the exception of nesting at Aves Island, Venezuela, while both sites in the eastern Atlantic demonstrated decreased nesting. These sites are not inclusive of all green sea turtle nesting in the Atlantic. However, other sites are not believed to support nesting levels high enough that would change the overall status of the species in the Atlantic (NMFS and USFWS 2007a).

By far, the most important nesting concentration for green sea turtles in the western Atlantic is in Tortuguero, Costa Rica (NMFS and USFWS 2007a). Nesting in the area has increased considerably since the 1970s and nest count data from 1999-2003 suggest nesting by 17,402-37,290 females per year (NMFS and USFWS 2007a). The number of females nesting per year on beaches in the Yucatán, at Aves Island, Galibi Reserve, and Isla Trindade number in the hundreds to low thousands, depending on the site (NMFS and USFWS 2007a).

The vast majority of green sea turtle nesting within the southeastern U.S. occurs in Florida (Johnson and Ehrhart 1994; Meylan et al. 1995). Green sea turtle nesting in Florida has been increasing since 1989 (Florida Fish and Wildlife Conservation Commission, Florida Marine Research Institute Index Nesting Beach Survey Database). Since establishment of index beaches in 1989, the pattern of green turtle nesting shows biennial peaks in abundance with a generally positive trend during the ten years of regular monitoring. This is perhaps due to increased protective legislation throughout the Caribbean (Meylan et al. 1995). A total statewide average (all beaches, including index beaches) of 5,039 green turtle nests were laid annually in Florida between 2001 and 2006, with a low of 581 in 2001 and a high of 9,644 in 2005 (NMFS and USFWS 2007a). Data from index nesting beaches substantiate the dramatic increase in nesting. In 2007, there were 9,455 green turtle nests found just on index nesting beaches, the highest since index beach monitoring began in 1989. The number fell back to 6,385 in 2008, further dropping under 3,000 in 2009, but that consecutive drop was a temporary deviation from the normal biennial nesting cycle for green turtles, as 2010 saw an increase back to 8,426 nests on the index nesting beaches (FWC Index Nesting Beach Survey Database). Occasional nesting has been documented along the Gulf coast of Florida (Meylan et al. 1995). More recently, green turtle nesting occurred on Bald Head Island, North Carolina; just east of the mouth of the Cape Fear River; on Onslow Island; and on Cape Hatteras National Seashore. In 2010, 18 nests were found in North Carolina, six nests in South Carolina, and six nests in Georgia (nesting databases maintained on www.seaturtle.org). Increased nesting has also been observed along the Atlantic coast of Florida, on beaches where only loggerhead nesting was observed in the past (Pritchard 1997). Recent modeling by Chaloupka et al. (2008a)using data sets of 25 years or more has resulted in an estimate of the Florida nesting stock at the Archie Carr National Wildlife Refuge growing at an annual rate of 13.9%, and the Tortuguero, Costa Rica, population growing at 4.9%.

There are no reliable estimates of the number of immature green sea turtles that inhabit coastal areas of the southeastern U.S. However, information on incidental captures of immature green sea turtles at the St. Lucie Power Plant in St. Lucie County, Florida, shows that the annual number of immature green sea turtles captured by their offshore cooling water intake structures has increased significantly. Green sea turtle annual captures averaged 19 for 1977-1986, 178 for 1987-1996, and 262 for 1997-2001 (Florida Power and Light Company St. Lucie Plant 2002). More recent unpublished data shows 101 captures in 2007, 299 in 2008, 38 in 2009 (power output was cut—and cooling water intake concomitantly reduced—for part of that year) and 413 in 2010. Ehrhart et al. (2007) documented a significant increase in in-water abundance of green turtles in the Indian River Lagoon area.

Indian Ocean. One of the largest nesting sites for green sea turtles worldwide occurs on the beaches of Oman where an estimated 20,000 green sea turtles nest annually (Hirth 1997a). Only the Comoros Island index site in the western Indian Ocean showed evidence of increased nesting (Seminoff 2004b).

Natural threats. Herons, gulls, dogfish, and sharks prey upon hatchlings. Adults face predation primarily by sharks and to a lesser extent by killer whales. All sea turtles except leatherbacks can undergo "cold stunning" if water temperatures drop below a threshold level, which can be lethal. For unknown reasons, the frequency of a disease called fibropapillomatosis is much higher in green sea turtles than in other species and threatens a large number of existing subpopulations. Extremely high incidence has been reported in Hawaii, where affliction rates peaked at 47-69% in some foraging areas (Murakawa et al. 2000). A to-date unidentified virus may aid in the development of fibropapillomatosis (Work et al. 2009). Predators (primarily of eggs and hatchlings) also include dogs, pigs, rats, crabs, sea birds, reef fishes, and groupers (Bell et al. 1994; Witzell 1981). Green sea turtles with an abundance of barnacles have been found to have a much greater probability of having health issues (Flint et al. 2009).

Anthropogenic threats. Major anthropogenic impacts to the nesting and marine environment affect green sea turtle survival and recovery. At nesting beaches, green sea turtles rely on intact dune structures, native vegetation, and normal beach temperatures for nesting (Ackerman 1997). Structural impacts to nesting habitat include the construction of buildings and pilings, beach armoring and renourishment, and sand extraction (Bouchard et al. 1998; Lutcavage et al. 1997). These factors may directly, through loss of beach habitat, or indirectly, through changing thermal profiles and increasing erosion, serve to decrease the amount of nesting area available to nesting females, and may evoke a change in the natural behaviors of adults and hatchlings (Ackerman 1997; Witherington et al. 2003; Witherington et al. 2007). The presence of lights on or adjacent to nesting beaches alters the behavior of nesting adults (Witherington 1992) and is often fatal to emerging hatchlings as they are attracted to light sources and drawn away from the water (Witherington and Bjorndal 1991). In addition to impacting the terrestrial zone, anthropogenic disturbances also threaten coastal marine habitats, particularly areas rich in seagrass and marine algae. These impacts include contamination from herbicides, pesticides, oil spills, and other chemicals, as well as structural degradation from excessive boat anchoring and dredging (Francour et al. 1999; Lee Long et al. 2000; Waycott et al. 2005). Ingestion of plastic and other marine debris is another source of morbidity and mortality (Stamper et al. 2009). Green sea turtles stranded in Brazil were all found to have ingested plastics or fishing debris (n=34), although mortality appears to have resulted in three cases (Tourinho et al. 2009). Low-level bycatch has also been documented in longline fisheries (Petersen et al. 2009). Further, the introduction of alien algae species threatens the stability of some coastal ecosystems and may lead to the elimination of preferred dietary species of green sea turtles (De Weede 1996). Very few green sea turtles are bycaught in U.S. fisheries (Finkbeiner et al. 2011). However, a legal fishery operates in Madagascar that harvested about 10,000 green turtles annually in the mid-1990s.

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

Harvest of sea turtle eggs or turtles themselves is a serious threat. Directed harvests of green sea turtle eggs and other life stages constitute a “major problem” in American Samoa, Guam, Palau, Commonwealth of the Northern Mariana Islands, Federated States of Micronesia, Republic of the Marshall Islands, and the Unincorporated Islands (Wake, Johnston, Kingman, Palmyra, Jarvis, Howland, Baker, and Midway). In the first half of the 20th century, 60,000 to 200,000 green sea turtle eggs were collected annually from beaches along Gangkou, China; free swimming individuals were heavily fished along the Xisha Archipelago (Frazier et al. 1988). Most nesting female green sea turtles were killed for their meat in Taiwan and overharvesting of hawksbills along China may be related to declines in abundance here as well (Chan et al. 2007a). Although Japan banned the importation of turtle shell in 1994, domestic harvests of eggs and turtles continue in the United States, its territories, and dependencies, particularly in the Caribbean and Pacific Island territories.

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

Critical habitat. On September 2, 1998, critical habitat for green sea turtles was designated in coastal waters surrounding Culebra Island, Puerto Rico (63 FR 46693). Aspects of these areas that are important for green sea turtle survival and recovery include important natal development habitat, refuge from predation, shelter between foraging periods, and food for green sea turtle prey.

Hawksbill sea turtle

Distribution. The hawksbill has a circumglobal distribution throughout tropical and, to a lesser extent, subtropical waters of the Atlantic, Indian, and Pacific oceans. Satellite tagged turtles have shown significant variation in movement and migration patterns. In the Caribbean, distance traveled between nesting and foraging locations ranges from a few kilometers to a few hundred kilometers (Byles and Swimmer 1994; Hillis-Starr et al. 2000; Horrocks et al. 2001; Lagueux et al. 2003; Miller et al. 1998; Prieto et al. 2001).

Population designation. Populations are distinguished generally by ocean basin and more specifically by nesting location. Our understanding of population structure is relatively poor. For example, genetic analysis of hawksbill sea turtles foraging off the Cape Verde Islands identified three closely-related haplotypes in a large majority of individuals sampled that did not match those of any known nesting population in the Western Atlantic, where the vast majority of nesting has been documented (McClellan et al. 2010; Monzon-Arguello et al. 2010).

Migration and movement. Upon first entering the sea, neonatal hawksbills in the Caribbean are believed to enter an oceanic phase that may involve long distance travel and eventual recruitment to nearshore foraging habitat (Boulon 1994). In the marine environment, the oceanic phase of juveniles (i.e., the "lost years") remains one of the most poorly understood aspects of hawksbill life history, both in terms of where turtles occur and how long they remain oceanic. Nesting site selection in the southwest Pacific appears to favor sites with higher wind and wave exposure, possibly as a means to aid hatchling dispersal (Garcon et al. 2010). Offshore waters associated with major current systems, such as the Kuroshio Extension, may be nursery habitat for hawksbills (NMFS and USFWS 1998c; Seminoff et al. 2003b). Adults may also be found in the Kuroshio Extension during interbreeding migrations (NMFS and USFWS 1998c; NMFS and USFWS 2007d).

Habitat. Hawksbill sea turtles are highly migratory and use a wide range of broadly separated localities and habitats during their lifetimes (Musick and Limpus 1997; Plotkin 2003). Small juvenile hawksbills (5-21 cm straight carapace length) have been found in association with *Sargassum* spp. in both the Atlantic and Pacific oceans (Musick and Limpus 1997) and observations of newly hatched hawksbills attracted to floating weed have been made (Hornell 1927; Mellgren and Mann 1996; Mellgren et al. 1994). Post-oceanic hawksbills may occupy a range of habitats that include coral reefs or other hard-bottom habitats, sea grass, algal beds, mangrove bays and creeks (Bjorndal and Bolten 2010; Musick and Limpus 1997), and mud flats (R. von Brandis, unpublished data in NMFS and USFWS 2007d). Individuals of multiple breeding locations can occupy the same foraging habitat (Bass 1999; Bowen et al. 1996; Bowen et al. 2007; Diaz-Fernandez et al. 1999; Velez-Zuazo et al. 2008). As larger juveniles, some individuals may associate with the same feeding locality for more than a decade, while others apparently migrate from one site to another (Blumenthal et al. 2009a; Mortimer et al. 2003; Musick and Limpus 1997). Larger individuals may prefer deeper habitats than their smaller counterparts (Blumenthal et al. 2009a). Nesting sites appear to be related to beaches with relatively high exposure to wind or wind-generated waves (Santana Garcon et al. 2010)

Occurrence in Japanese waters is infrequent, but individuals are known from both coasts and as far north as 38º N and likely involve individuals entrained in the Kuroshio Current (Kamezaki and Matsui 1997; Uchida 1994; Uchida and Nishiwaki 1995).

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

The life history of hawksbills consists of a pelagic stage that lasts from hatching until they are approximately 22-25 cm in straight carapace length (Meylan 1988; Meylan and Donnelly 1999), followed by residency in coastal developmental habitats.

Feeding. Dietary data from oceanic stage hawksbills are limited, but indicate a combination of plant and animal material (Bjorndal 1997).

Diving. Hawksbill diving ability varies with age and body size. As individual size increases with age, diving ability in terms of duration and depth increases (Blumenthal et al. 2009b). Studies of hawksbills in the Caribbean have found diurnal diving behavior, with dive duration nearly twice as long during nighttime (35-47 min) compared to daytime (19-26 min Blumenthal et al. 2009b; Van Dam and Diez 1997). Daytime dives averaged 5 m, while nighttime dives averaged 43 m (Blumenthal et al. 2009b)

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

Vocalization and hearing. Although information is not available regarding hawksbill sea turtle vocalizations or auditory capabilities, green and loggerhead sea turtles have been studied and are likely similar in capacity to their close relative, the hawksbill. The frequency range at which these species hear best is 200-400 Hz, with rapid diminishment of sensitivity outside of this range. Green and loggerhead sea turtles are likely incapable of hearing frequencies >1,000 Hz (Moein Bartol et al. 1999; Ridgway et al. 1969).

Status and trends. Hawksbill sea turtles received protection on June 2, 1970 (35 FR 8495) under the Endangered Species Conservation Act and since 1973 have been listed as endangered under the ESA. Although no historical records of abundance are known, hawksbill sea turtles are considered to be severely depleted due to the fragmentation and low use of current nesting beaches (NMFS and USFWS 2007d). Worldwide, an estimated 21,212-28,138 hawksbills nest each year among 83 sites. Among the 58 sites with historic trends, all show a decline during the past 20 to 100 years. Among 42 sites for which recent trend data are available, 10 (24%) are increasing, three (7%) are stable and 29 (69%) are decreasing. Encouragingly, nesting range along Mexico and Central America appears not to have contracted and estimates continue to increase as additional dedicated study is conducted in the eastern Pacific (Gaos et al. 2010a).

Atlantic Ocean. Atlantic nesting sites include: Antigua (Jumby Bay), the Turks and Caicos, Barbados, the Bahamas, Puerto Rico (Mona Island), the U.S. Virgin Islands, the Dominican Republic, Sao Tome, Guadaloupe, Trinidad and Tobago, Jamaica, Martinique, Cuba (Doce Leguas Cays), Mexico (Yucatan Peninsula), Costa Rica (Tortuguero National Park), Guatemala, Venezuela, Bijagos Archipelago, Guinea-Bissau, and Brazil.

Population increase has been greater in the Insular Caribbean than along the Western Caribbean Mainland or the eastern Atlantic (including Sao Tomé and Equatorial Guinea). Nesting populations of Puerto Rico appeared to be in decline until the early 1990s, but have universally increased during the survey periods. Mona Island now hosts 199-332 nesting females annually, and the other sites combined host 51-85 nesting females annually (R.P. van Dam and C.E. Diez, unpublished data in NMFS and USFWS 2007d) C.E. Diez, Chelonia, Inc., in litt. to J. Mortimer

2006). The U.S. Virgin Islands have a long history of tortoiseshell trade (Schmidt 1916). At Buck Island Reef National Monument, protection has been in force since 1988, and during that time, hawksbill nesting has increased by 143% to 56 nesting females annually, with apparent spill over to beaches on adjacent St. Croix (Z. Hillis-Starr, National Park Service, in litt. to J. Mortimer 2006). However, St. John populations did not increase, perhaps due to the proximity of the legal turtle harvest in the British Virgin Islands (Z. Hillis-Starr, National Park Service, in litt. to J. Mortimer 2006). Populations have also been identified in Belize and Brazil as genetically unique (Hutchinson and Dutton 2007). An estimated 50-200 nests are laid per year in the Guinea-Bissau (Catry et al. 2009).

Pacific Ocean. American Samoa and Western Samoa host fewer than 30 females annually (Grant et al. 1997; Tuato'o-Bartley et al. 1993). In Guam, only 5-10 females are estimated to nest annually (G. Balazs, NMFS, in litt. to J. Mortimer 2007; G. Davis, NMFS, in litt. to J. Mortimer 2007) and the same is true for Hawaii, but there are indications that this population is increasing (G. Balazs, pers. comm. in NMFS and USFWS 2007d). Additional populations are known from the eastern Pacific (potentially extending from Mexico through Panama), northeastern Australia, and Malaysia (Hutchinson and Dutton 2007). El Salvador is now known to host the majority of hawksbill turtle nesting activity in the eastern Pacific, with 79.6% (n= 5430) of all nesting observation records, and Mexico hosting the majority of records of hawksbill turtles at sea, with 60.3% (n= 544) of all in-water observation records (Gaos et al. 2010b). Total number of nesting females for the Central Pacific hawksbill population was estimated at 940 – 1,200 females annually for the last few years, with an overall downward trend (NMFS and USFWS 2007b).

Indian Ocean. The Indian Ocean hosts several populations of hawksbill sea turtles (Hutchinson and Dutton 2007; Spotila 2004b). These include western Australian, Andaman and Nicobar islands, Maldives, Seychelles, Burma, East Africa, Egypt, Oman, Saudi Arabia, Sudan, and Yemen.

Natural threats. Sea turtles face predation primarily by sharks and to a lesser extent by killer whales. All sea turtles except leatherbacks can undergo “cold stunning” if water temperatures drop below a threshold level, which can be lethal. The only other significant natural threat to hawksbill sea turtles is from hybridization of hawksbills with other species of sea turtles. This is especially problematic at certain sites where hawksbill numbers are particularly low (Mortimer and Donnelly in review). Predators (primarily of eggs and hatchlings) include dogs, pigs, rats, crabs, sea birds, reef fishes, groupers, feral cats, and foxes (Bell et al. 1994; Ficetola 2008). In some areas, nesting beaches can be almost completely destroyed and all nests can sustain some level of depredation (Ficetola 2008).

Anthropogenic threats. Threats to hawksbill sea turtles are largely anthropogenic, both historically and currently. Impacts to nesting beaches include the construction of buildings and pilings, beach armoring and renourishment, and sand extraction (Bouchard et al. 1998; Lutcavage et al. 1997). Because hawksbills prefer to nest under vegetation (Horrocks and Scott 1991; Mortimer 1982), they are particularly impacted by beachfront development and clearing of dune vegetation (Mortimer and Donnelly in review). The presence of lights on or adjacent to nesting beaches alters the behavior of nesting adults (Witherington 1992) and is often fatal to emerging hatchlings as they are attracted to light sources and drawn away from the water (Witherington and Bjorndal 1991). One of the most detrimental human threats to hawksbill sea turtles is the intensive harvest of eggs from nesting beaches. Between 1950 and 1992, approximately 1.3 million hawksbill shells were collected to supply tortoiseshell to the Japanese market, the world's largest. Japan stopped importing tortoiseshell in 1993 in order to comply with CITES (Limpus and Miller 2008).

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

Large numbers of nesting and foraging hawksbill sea turtles are captured and killed for trade in Micronesia, the Mexican Pacific coast, southeast Asia, and Indonesia (NMFS and USFWS 1998c). In addition to the demand for the hawksbill's shell, there is a demand for other products including leather, oil, perfume, and cosmetics. Before the U.S. certified Japan under the Pelly Amendment, Japan had been importing about 20 metric tons of hawksbill shell per year, representing approximately 19,000 turtles.

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

Critical habitat. On September 2, 1998, the NMFS established critical habitat for hawksbill sea turtles around Mona and Monito Islands, Puerto Rico (63 FR 46693). Aspects of these areas that are important for hawksbill sea turtle survival and recovery include important natal development habitat, refuge from predation, shelter between foraging periods, and food for hawksbill sea turtle prey.

Leatherback sea turtle

Distribution. Leatherbacks range farther than any other sea turtle species, having evolved physiological and anatomical adaptations that allow them to exploit cold waters (Frair et al. 1972; Greer et al. 1973; USFWS 1995). High-latitude leatherback range includes in the Atlantic includes the North and Barents Seas, Newfoundland and Labrador , Argentina, and South Africa (Goff and Lien 1988; Hughes et al. 1998; Luschi et al. 2003; Luschi et al. 2006; Márquez 1990; Threlfall 1978). Pacific ranges extend to Alaska, Chile, and New Zealand (Brito 1998; Gill 1997; Hodge and Wing 2000).

Leatherbacks also occur in Mediterranean and Indian Ocean waters (Casale et al. 2003; Hamann et al. 2006a). Associations exist with continental shelf and pelagic environments and sightings occur in offshore waters of 7-27˚ C (CETAP 1982). Juvenile leatherbacks usually stay in warmer, tropical waters >21˚ C (Eckert 2002). Males and females show some degree of natal homing to annual breeding sites (James et al. 2005).

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

Atlantic Ocean. Previous genetic analyses of leatherbacks using only mitochondrial DNA (mtDNA) resulted in an earlier determination that within the Atlantic basin there are at least three genetically different nesting populations: the St. Croix nesting population (U.S. Virgin Islands), the mainland nesting Caribbean population (Florida, Costa Rica, Suriname/French Guiana), and the Trinidad nesting population (Dutton et al. 1999). Further genetic analyses using microsatellite markers in nuclear DNA along with the mtDNA data and tagging data has resulted in Atlantic Ocean leatherbacks now being divided into seven groups or breeding populations: Florida, Northern Caribbean, Western Caribbean, Southern Caribbean/Guyana's, West Africa, South Africa, and Brazil (TEWG 2007a).

Caribbean Sea. Nesting occurs in Puerto Rico, St. Croix, Costa Rica, Panama, Colombia, Trinidad and Tobago, Guyana, Suriname, and French Guiana (Bräutigam and Eckert 2006; Márquez 1990; Spotila et al. 1996).

Indian Ocean. Nesting is reported in South Africa, India, Sri Lanka, and the Andaman and Nicobar islands(Hamann et al. 2006a).

Pacific Ocean. Leatherbacks are found from tropical waters north to Alaska within the North Pacific and is the most common sea turtle in the eastern Pacific north of Mexico (Eckert 1993a; Stinson 1984a; Wing and Hodge 2002). The west coast of Central America and Mexico hosts nesting from September-March, although Costa Rican nesting peaks during April-May (Chacón-Chaverri and Eckert 2007; LGL Ltd. 2007). Leatherback nesting aggregations occur widely in the Pacific, including China, Malaysia, Papua New Guinea, Indonesia, Thailand, Australia, Fiji, the Solomon Islands, and Central America (Dutton et al. 2007; Limpus 2002). Significant nesting also occurs along the Central American coast (Márquez 1990). Although not generally known to nest on Japanese shores, two nests were identified in the central Ryukyu Islands in 2002 (Kamezaki et al. 2002).

In Costa Rica, leatherbacks nest at Playa Naranjo in Santa Rosa National Park, the second-most important nesting beach on the Pacific coast (Yañez et al. 2010), Rio Oro on the Osa Peninsula, and at various beaches in Las Baulas National Park, which includes Playa Langosta and Playa Grande and contains the largest colony of leatherbacks in the Pacific (Spotila 2004b). Females typically lay six clutches per season (average nine days between nests), which incubate for 58–65 days (Lux et al. 2003). Limited nesting also occurs along Nicaragua, Panama, El Salvador, Vanuatu, and Guatemala.

Growth and reproduction. It has been thought that leatherbacks reach sexual maturity somewhat faster than other sea turtles (except Kemp's ridley), with an estimated range of 3-6 years (Rhodin 1985) to 13-14 years (Zug and Parham 1996). However, recent research suggests otherwise, with western North Atlantic leatherbacks possibly not maturing until as late as 29 years of age (Avens and Goshe 2007). Female leatherbacks nest frequently (up to 10 nests per year and about every 2-3 years). During each nesting, females produce 100 eggs or more per clutch and 700 eggs or more per nesting season (Schultz 1975). However, up to ~30% of the eggs can be infertile. Thus, the actual proportion of eggs that can result in hatchlings is less than this seasonal estimate. The eggs incubate for 55-75 days before hatching.

Habitat. Leatherbacks occur throughout marine waters, from nearshore habitats to oceanic environments (Grant and Ferrell 1993; Schroeder and Thompson 1987; Shoop and Kenney 1992; Starbird et al. 1993). Movements are largely dependent upon reproductive and feeding cycles and the oceanographic features that concentrate prey, such as frontal systems, eddy features, current boundaries, and coastal retention areas (Benson et al. 2011; Collard 1990; Davenport and Balazs 1991; Frazier 2001; HDLNR 2002). Aerial surveys off the western U.S. support continental slope waters as having greater leatherback occurrence than shelf waters (Bowlby et al. 1994; Carretta and Forney 1993; Green et al. 1992; Green et al. 1993). Nesting sites appear to be related to beaches with relatively high exposure to wind or wind-generated waves (Santana Garcon et al. 2010).

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

Migration and movement. Leatherback sea turtles migrate throughout open ocean convergence zones and upwelling areas, along continental margins, and in archipelagic waters (Eckert 1998; Eckert 1999; Morreale et al. 1994). In a single year, a leatherback may swim more than 9,600 km to nesting and foraging areas throughout ocean basins (Benson et al. 2007a; Benson et al. 2007b; Eckert 1998; Eckert 2006; Eckert et al. 2006; Ferraroli et al. 2004; Hays et al. 2004; Sale et al. 2006). Much of this travel may be due to movements within current and eddy features, moving individuals along (Sale and Luschi 2009). Return to nesting beaches may be accomplished by a form of geomagnetic navigation and use of local cues (Sale and Luschi 2009). Leatherback females will either remain in nearshore waters between nesting events, or range widely, presumably to feed on available prey (Byrne et al. 2009; Fossette et al. 2009a). Leatherbacks are known to pass by the Japanese coast during developmental, foraging, and reproductive migrations (Sea Turtle Association of Japan 2010; Uchida and Nishiwaki 1995).

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

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

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

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

Feeding. Leatherbacks may forage in high-invertebrate prey density areas formed by favorable features (Eckert 2006; Ferraroli et al. 2004). Although leatherbacks forage in coastal waters, they appear to remain primarily pelagic through all life stages (Heppell et al. 2003). The location and abundance of prey, including medusae, siphonophores, and salpae, in temperate and boreal latitudes likely has a strong influence on leatherback distribution in these areas (Plotkin 1995). Leatherback prey are frequently found in the deep-scattering layer in the Gulf of Alaska (Hodge and Wing 2000). North Pacific foraging grounds contain individuals from both eastern and western Pacific rookeries, although leatherbacks from the eastern Pacific generally forage in the Southern Hemisphere along Peru and Chile (Dutton 2005-2006; Dutton et al. 2000; Dutton et al. 1998). Mean primary productivity in all foraging areas of western Atlantic females is 150% greater than in eastern Pacific waters, likely resulting in twice the reproductive output of eastern Pacific females (Saba et al. 2007). Leatherbacks have been observed feeding on jellyfish in waters off Washington State and Oregon (Eisenberg and Frazier 1983; Stinson 1984a). Although leatherback sea turtles have not been caught as longline bycatch nor have they been otherwise observed in the Shatsky Rise area, the presence of frontal boundaries and potential feeding habitat suggests that this species may be present here (Komatsu et al. 2002; Polovina et al. 2000; Yokota et al. 2006).

Diving. Leatherbacks are champion deep divers among sea turtles with a maximum- recorded dive of over 4,000 m (Eckert et al. 1989; López-Mendilaharsu et al. 2009). Dives are typically 50-84 m and 75-90% of time duration is above 80 m (Standora et al. 1984). Leatherbacks off South Africa were found to spend <1% of their dive time at depths greater than 200 m (Hays et al. 2009). Dive durations are impressive, topping 86 min, but routinely 1-14 min (Eckert et al. 1989; Eckert et al. 1996; Harvey et al. 2006; López-Mendilaharsu et al. 2009). Most of this time is spent traveling to and from maximum depths (Eckert et al. 1989). Dives are continual, with only short stays at the surface (Eckert et al. 1989; Eckert et al. 1986; Southwood et al. 1999). Off Playa Grande, Costa Rica, adult females spent 57–68% of their time underwater, diving to a mean depth of 19 m for 7.4 min (Southwood et al. 1999). Off St. Croix, adult females dove to a mean depth of 61.6 m for an average of 9.9 min, and spent an average of 4.9 min at the surface (Eckert et al. 1989). During shallow dives in the South China Sea, dives averaged 6.9–14.5 min, with a maximum of 42 min (Eckert et al. 1996). Off central California, leatherbacks dove to 20–30 m with a maximum of 92 m (Harvey et al. 2006). This corresponded to the vertical distribution of their prey (Harvey et al. 2006). Leatherback prey in the Gulf of Alaska are frequently concentrated in the deep-scattering layer (Hodge and Wing 2000). Mean dive and surface durations were 2.9 and 2.2 min, respectively (Harvey et al. 2006). In a study comparing diving patterns during foraging versus travelling, leatherbacks dove shallower (mean of 53.6 m) and moved more slowly (17.2 km/day) while in foraging areas while travelling to or from these areas (81.8 m and 51.0 km/day) (Fossette et al. 2009b).

Vocalization and hearing. Information on the hearing capabilities of sea turtles is limited, but the information that is available suggests auditory capabilities are centered in the low-frequency range (< 1 kHz), with hearing thresholds at about 132-140 dB (Lenhardt 1994; Lenhardt et al. 1983; Moein Bartol and Ketten 2006; Moein Bartol et al. 1999; O'Hara and Wilcox 1990; Ridgway et al. 1969). There is some sensitivity to frequencies as low as 60 Hz, and probably as low as 30 Hz (L-DEO 2006).

Status and trends. Leatherback sea turtles received protection on June 2, 1970 (35 FR 8491) under the Endangered Species Conservation Act and, since 1973, have been listed as endangered under the ESA, but declines in nesting have continued worldwide. Breeding females were initially estimated at 29,000-40,000, but were later refined to ~115,000 (Pritchard 1971; Pritchard 1982). Spotila et al. (1996) estimated 34,500 females, but later issued an update of 35,860 (Spotila 2004a). The species as a whole is declining and local populations are in danger of extinction (NMFS 2001a).

Nesting aggregations occur along Gabon, Sao Tome and Principe, French Guiana, Suriname, and Florida (Bräutigam and Eckert 2006; Márquez 1990; Spotila et al. 1996). Widely dispersed but fairly regular African nesting also occurs between Mauritania and Angola (Fretey et al. 2007). Many sizeable populations (perhaps up to 20,000 females annually) of leatherbacks are known to nest in West Africa (Fretey 2001). The population of leatherbacks nesting on Gabon beaches has been suggested as being the world’s largest, with 36,185-126,480 clutches being laid by 5,865-20,499 females annually from 2002-2007 (Witt et al. 2009). The total number of females utilizing Gabon nesting beaches is estimated to be 15,730- 41,373 (Witt et al. 2009). North Atlantic leatherbacks likely number 34,000-94,000 individuals, with females numbering 18,800 and the eastern Atlantic segment numbering 4,700 (TEWG 2007a). Trends and numbers include only nesting females and are not a complete demographic or geographic cross-section. In 1996, the entire Western Atlantic population was characterized as stable at best (Spotila et al. 1996), with roughly 18,800 nesting females. A subsequent analysis indicated that by 2000, the western Atlantic nesting population had decreased to about 15,000 nesting females (NMFS 2011). Spotila et al. (1996) estimated that the entire Atlantic basin, including all nesting beaches in the Americas, the Caribbean, and West Africa, totaled approximately 27,600 nesting females, with an estimated range of 20,082-35,133. This is consistent with other estimates of 34,000-95,000 total adults (20,000-56,000 adult females; 10,000-21,000 nesting females)(TEWG 2007b).

The largest nesting aggregation in the western North Atlantic occurs in French Guiana and Suriname, likely belongs to a metapopulation whose limits remain unknown (Rivalan et al. 2006). Heppell et al. (2003a) concluded that leatherbacks generally show less genetic structuring than green and hawksbill sea turtles. The French Guiana nesting aggregation has declined ~15%

annually since 1987 (NMFS 2001a). However, from 1979-1986, the number of nests increased ~15% annually, possibly indicating the current decline may be linked with the erosion cycle of Guiana beaches (NMFS 2006e). Guiana nesting may have increased again in the early 2000s (NMFS 2006e). Suriname nesting numbers have recently increased from more than 10,000 nests annually since 1999 and a peak of 30,000 nests in 2001. Overall, Suriname and French Guiana nesting trends towards an increase (Girondot et al. 2007; Hilterman and Goverse 2003). Florida (March-July) and U.S. Caribbean nesting since the early 1980s has increased ~0.3% and 7.5% per year, respectively, but lags behind the French Guiana coast and elsewhere in magnitude (NMFS/SEFSC 2001). This positive growth was seen within major nesting areas for the stock, including Trinidad, Guyana, and the combined beaches of Suriname and French Guiana (TEWG 2007b). Using both Bayesian modeling and regression analyses, the TEWG (2007b) determined that the Southern Caribbean/Guyanas stock had demonstrated a long-term, positive population growth rate (using nesting females as a proxy for population).

The Caribbean coast of Costa Rica and extending through Chiriquí Beach, Panama, represents the fourth largest known leatherback rookery in the world (Troeng et al. 2004). Examination of data from three index nesting beaches in the region (Tortuguero, Gandoca, and Pacuare in Costa Rica) using various Bayesian and regression analyses indicated that the nesting population likely was not growing during 1995-2005 (TEWG 2007b). Other modeling of the nesting data for Tortuguero indicates a 67.8% decline between 1995 and 2006 (Troëng et al. 2007).

In Puerto Rico, the primary nesting beaches are at Fajardo and on the island of Culebra. Nesting between 1978 and 2005 ranged between 469-882 nests, and the population has been growing since 1978, with an overall annual growth rate of 1.1% (TEWG 2007b). At the primary nesting beach on St. Croix, the Sandy Point National Wildlife Refuge, nesting has fluctuated from a few hundred nests to a high of 1,008 in 2001, and the average annual growth rate has been approximately 1.1% from 1986-2004 (TEWG 2007b).

The Florida nesting stock comes ashore primarily along the east coast of Florida. This stock is of growing importance, with total nests between 800-900 per year in the 2000s following nesting totals fewer than 100 nests per year in the 1980s (NMFS 2011). Using data from the index nesting beach surveys, the TEWG (2007b) estimated a significant annual nesting growth rate of 1% between 1989 and 2005. Stewart et al. (2011) evaluated nest counts from 68 Florida beaches over 30 years (1979-2008) and found that nesting increased at all beaches with trends ranging from 3.1%-16.3% per year, with an overall increase of 10.2% per year. In 2007, a record 517 leatherback nests were observed on the index beaches in Florida, with 265 in 2008, and then an increase to a new record of 615 nests in 2009, and a slight decline in 2010 back to 552 nests (FWC Index Nesting Beach database). This up-and-down pattern is thought to be a result of the cyclical nature of leatherback nesting, similar to the biennial cycle of green turtle nesting.

The most recent population estimate for leatherback sea turtles from the North Atlantic as a whole is between 34,000-90,000 adult individuals (20,000-56,000 adult females)(TEWG 2007b).

Heavy declines have occurred at all major Pacific basin rookeries, as well as Mexico, Costa Rica, Malaysia, India, Sri Lanka, Thailand, Trinidad, Tobago, and Papua New Guinea. This includes a nesting decline of 23% between 1984-1996 at Mexiquillo, Michoacán, Mexico (Sarti et al. 1996). According to reports from the late 1970s and early 1980s, three beaches on the Pacific coast of Mexico supported as many as half of all leatherback turtle nests for the eastern Pacific. Since the early 1980s, the eastern Pacific Mexican population of adult female leatherback turtles has declined to slightly more than 200 individuals during 1998-1999 and 1999-2000 (Sarti et al. 2000). Spotila et al. (2000) reported the decline of the leatherback turtle population at Playa Grande, Costa Rica, which had been the fourth largest nesting colony in the world. Between 1988 and 1999, the nesting colony declined from 1,367 to 117 female leatherback turtles. Based on their models, Spotila et al. (2000)estimated that the colony could fall to less than 50 females by 2003-2004. Fewer than 1,000 females nested on the Pacific coast of Mexico from 1995-1996 and fewer than 700 females are estimated for Central America (Spotila et al. 2000). The number of leatherback turtles nesting in Las Baulas National Park declined rapidly during the 1990s, from about 1,500 females during the 1988–89 nesting season, to about 800 in 1990–91 and 1991–92 to 193 in 1993–94 (Williams et al. 1996) and 117 in 1998–99 (Spotila et al. 2000). Spotila (2004b) reported that between 59 and 435 leatherbacks nest at Las Baulas each year depending on the El Niño–La Niña cycle. Only an Indonesian nesting assemblage has remained relatively abundant in the Pacific basin. The largest extant leatherback nesting assemblage in the Indo-Pacific lies on the northern Vogelkop coast of Irian Jaya (West Papua), Indonesia, with roughly 3,000 nests recorded annually (Dutton et al. 2007; Putrawidjaja 2000; Suárez et al. 2000). The Western Pacific leatherback metapopulation harbors the last remaining nesting aggregation of significant size in the Pacific with approximately 2700–4500 breeding females (Dutton et al. 2007; Hitipeuw et al. 2007). The total number of nests per year for the Jamursba-Medi leatherback nesting population ranged between a high of 6,373 nests in 1996 and a low of 1,537 nests in 2010 (Hitipeuw et al. 2007).

Declines in the western Pacific are equally severe. Nesting at Terengganu, Malaysia is 1% of that in the 1950s (Chan and Liew 1996). The South China Sea and East Pacific nesting colonies have undergone catastrophic collapse. Overall, Pacific populations have declined from an estimated 81,000 individuals to <3,000 total adults and subadults (Spotila et al. 2000). The number of nesting leatherbacks has declined by an estimated 95% over the past 20 years in the Pacific (Gilman 2009). Drastic overharvesting of eggs and mortality from fishing activities is likely responsible for this tremendous decline (Eckert 1997; Sarti et al. 1996).

Based on the survey and tagging work, it was estimated that 400-500 female leatherbacks nest annually on Great Nicobar Island (Andrews et al. 2002). The number of nesting females using the Andaman and Nicobar Islands combined was estimated around 1,000 (Andrews and Shanker 2002).

Natural threats. Sea turtles face predation primarily by sharks and to a lesser extent by killer whales (Pitman and Dutton 2004). Hatchlings are preyed upon by herons, gulls, dogfish, and sharks. Leatherback hatching success is particularly sensitive to nesting site selection, as nests that are overwashed have significantly lower hatching success and leatherbacks nest closer to the high-tide line than other sea turtle species (Caut et al. 2009a).

Anthropogenic threats. Leatherback nesting and marine environments are facing increasing impacts through widespread development and tourism along nesting beaches (Hamann et al. 2006a; Hernandez et al. 2007; Maison 2006; Santidrián Tomillo et al. 2007). Structural impacts to beaches include building and piling construction, beach armoring and renourishment, and sand extraction (Bouchard et al. 1998; Lutcavage et al. 1997). In some areas, timber and marine debris accumulation as well as sand mining reduce available nesting habitat (Bourgeois et al. 2009; Chacón Chaverri 1999; Formia et al. 2003; Laurance et al. 2008). Lights on or adjacent to nesting beaches alter nesting adult behavior and is often fatal to emerging hatchlings as they are drawn to light sources and away from the sea (Bourgeois et al. 2009; Cowan et al. 2002; Deem et al. 2007; Witherington 1992; Witherington and Bjorndal 1991). Plastic ingestion is very common in leatherbacks and can block gastrointestinal tracts leading to death (Mrosovsky et al. 2009). Although global warming may expand foraging habitats into higher latitude waters, increasing temperatures may increase feminization of nests (Hawkes et al. 2007b; James et al. 2006; McMahon and Hays 2006; Mrosovsky et al. 1984). Rising sea levels may also inundate nests on some beaches. Egg collection is widespread and attributed to catastrophic declines, such as in Malaysia. Harvest of females along nesting beaches is of concern worldwide.

Bycatch, particularly by longline fisheries, is a major source of mortality for leatherback sea turtles (Crognale et al. 2008; Fossette et al. 2009a; Gless et al. 2008; Petersen et al. 2009). Wallace et al. (2010) estimated that between 1990 and 2008, at least 85,000 sea turtles were captured as bycatch in fisheries worldwide. This estimate is likely at least two orders of magnitude low, resulting in a likely bycatch of nearly half a million sea turtles annually (Wallace et al. 2010); many of these turtles are expected to be leatherbacks. Japanese research and training vessels are estimated to have bycaught large numbers of sea turtles historically (21,200 bycaught, 12,296 killed) (Nishimura and Nakahigashi 1990). An estimated 20,000-40,000 leatherbacks were captured as longline bycatch in the Pacific during 2000, with 1,000-1,300 of these dying (Lewison et al. 2004b). Shallow-set longline fisheries based out of Hawaii likely captured and killed several hundred leatherback sea turtles before their closure in 2001. When fisheries re-opened in 2004, with substantial modifications to protect sea turtles, estimates of leatherbacks captured and killed dropped to one or two each year. Between 2004 and 2008, shallow-set fisheries based out of Hawaii captured about 19 leatherbacks, killing about five. In a biological opinion released in October 2000, NMFS concluded that the California/Oregon drift gillnet fishery, which operates mostly along the California coast, would jeopardize leatherback and loggerhead sea turtles (NMFS 2000). From 15 August-15 November every year, mesh gill nets larger than 14 inches are prohibited between Point Conception, California and northern Oregon (45ºN) as a mechanism to reduce sea turtle mortality (50 CFR 660.713(c)). Leatherback sea turtles have also been and are likely continue to be captured and killed in the deep-set based longline fisheries based out of Hawaii and American Samoa. 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 occurred to the East Pacific population). Spotila (2000) asserts that most of the mortality associated with the Playa Grande nesting site was fishery related. Lewison et al. (2009) analyzed bycatch of several marine mammal, avian, and sea turtle species in the North Pacific Ocean and found several persistent bycatch hotspots, with bycatch occurring in clusters at relatively high levels for the number of fishing sets in the area: off the California and Mexican coast, along the northwestern Hawaiian Islands, and in a region of the north central Pacific above 40º N.

Donoso and Dutton (2010) found that 284 leatherbacks were bycaught between 2001 and 2005 as part of the Chilean longline fishery, with two individuals observed dead; leatherbacks were the most frequently bycaught sea turtle species. Between 8-17 leatherback turtles likely died annually between 1990 and 2000 in interactions with the California/Oregon drift gillnet fishery; 500 leatherback turtles are estimated to die annually in Chilean and Peruvian fisheries; 200 leatherback turtles are estimated to die in direct harvests in Indonesia; and, before 1992, the North Pacific driftnet fisheries for squid, tuna, and billfish captured an estimated 1,000 leatherback turtles each year, killing about 111 of them each year. Currently, the U.S. tuna and swordfish longline fisheries managed under the HMS FMP are estimated to capture 1,764 leatherbacks (no more than 252 mortalities) for each 3-year period starting in 2007 (NMFS 2004a). All leatherbacks were released alive, with all gear removed for the majority of captures. Lewison et al. (2004a) estimated that 30,000-60,000 leatherbacks were taken in all Atlantic longline fisheries in 2000 (including the U.S. Atlantic tuna and swordfish longline fisheries, as well as others). Finkbeiner et al. (2011) estimated that annual bycatch interactions total 1,400 individuals annually for U.S. Atlantic fisheries (resulting in roughly forty mortalities) and one hundred interactions in U.S. Pacific fisheries (resulting in about ten mortalities). Mortality of leatherbacks in the U.S. shrimp fishery is now estimated at 54 turtles per year. Data collected by the NEFSC Fisheries Observer Program from 1994 through 1998 (excluding 1997) indicate that 37 leatherbacks were incidentally captured (16 lethally) in drift gillnets set in offshore waters from Maine to Florida during this period. Observer coverage for this period ranged from 54 to 92%. Trinidad and Tobago's Institute for Marine Affairs estimated that more than 3,000 leatherbacks were captured incidental to gillnet fishing in the coastal waters of Trinidad in 2000. Half or more of the gravid turtles in Trinidad and Tobago waters may be killed (Lee Lum 2003), though many of the turtles do not die as a result of drowning, but rather because the fishermen butcher them in order to get them out of their nets (NMFS 2001b).

We know little about the effects of contaminants on leatherback sea turtles. The metals arsenic, cadmium, copper, mercury, selenium, and zinc bioaccumulate, with cadmium in highest concentration in leatherbacks versus any other marine vertebrate (Caurant et al. 1999; Gordon et al. 1998). A diet of primarily jellyfish, which have high cadmium concentrations, is likely the cause (Caurant et al. 1999). Organochlorine pesticides have also been found (Mckenzie et al. 1999). PCB concentrations are reportedly equivalent to those in some marine mammals, with liver and adipose levels of at least one congener being exceptionally high (PCB 209: 500-530 ng/g wet weight Davenport et al. 1990; Oros et al. 2009).

Critical habitat. On March 23, 1979, leatherback critical habitat was identified adjacent to Sandy Point, St. Croix, U.S.V.I. from the 183 m isobath to mean high tide level between 17° 42’12” N and 65°50’00” W (44 FR 17710). This habitat is essential for nesting, which has been increasingly threatened since 1979, when tourism increased significantly, bringing nesting habitat and people into close and frequent proximity. However, studies do not currently support significant critical habitat deterioration.

On January 26, 2012, the NMFS designated critical habitat for leatherback sea turtles in waters along Washington State (Cape Flattery to Cape Blanco; 64,760 km^2) and California (Point Arena to Point Arguello; 43,798 km^2). The primary constituent elements of these areas include (1.) the occurrence of prey species, primarily scyphomedusae of the order Semaeostomeae (*Chrysaora, Aurelia, Phacellophora*, and *Cyanea*) of sufficient condition, distribution, diversity, and abundance to support individual as well as population growth, reproduction, and development and (2) migratory pathway conditions to allow for safe and timely passage and access to/from/within high use foraging areas.

Loggerhead Sea Turtle-North Pacific DPS

Distribution. Loggerheads are circumglobal occurring throughout the temperate and tropical regions.

Population designations. As with other sea turtles, populations are frequently divided by nesting aggregation (Hutchinson and Dutton 2007). On September 22, 2011, the NMFS designated nine distinct population segments (DPSs) of loggerhead sea turtles: South Atlantic Ocean and southwest Indian Ocean as threatened as well as Mediterranean Sea, North Indian Ocean, North Pacific Ocean, northeast Atlantic Ocean, northwest Atlantic Ocean, South Pacific Ocean, and southeast Indo-Pacific Ocean as endangered (75 FR 12598). We expect only the North Pacific DPS would be exposed during the proposed activities.

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

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

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

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

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

Migration and movement. Loggerhead hatchlings migrate offshore and become associated with *Sargassum* spp. habitats, driftlines, and other convergence zones (Carr 1986). After 14-32 years of age, they shift to a benthic habitat, where immature individuals forage in the open ocean and coastal areas along continental shelves, bays, lagoons, and estuaries (Bowen et al. 2004; NMFS 2001b). Adult loggerheads make lengthy migrations from nesting beaches to foraging grounds (TEWG 1998).

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

Gender, age, and survivorship. Although information on males is limited, several studies identified a female bias, although a single study has found a strong male bias to be possible (Dodd 1988; NMFS 2001b; Rees and Margaritoulis 2004).

Additionally, little is known about longevity, although Dodd (1988) estimated the maximum female life span at 47-62 years. Heppell et al. (2003) estimated annual survivorship to be 0.81 (southeast U.S. adult females), 0.78-0.91 (Australia adult females), 0.68-0.89 (southeast U.S. benthic juveniles, and 0.92 (Australia benthic juveniles). Survival rates for hatchlings during their first year are likely very low (Heppell et al. 2003).

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

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

Vocalization and hearing. Information on the hearing capabilities of sea turtles is limited, but available information suggests auditory capabilities are centered in the low-frequency range (< 1 kHz), with hearing thresholds at about 132-140 dB (Lenhardt 1994; Lenhardt et al. 1983; Moein Bartol and Ketten 2006; Moein Bartol et al. 1999; O'Hara and Wilcox 1990; Ridgway et al. 1969). There is some sensitivity to frequencies as low as 60 Hz, and probably as low as 30 Hz (L-DEO 2006).

Status and trends. Loggerhead sea turtles were listed as threatened under the ESA of 1973 on July 28, 1978 (43 FR 32800). There is general agreement that the number of nesting females provides a useful index of the species' population size and stability at this life stage, even though there are doubts about the ability to estimate the overall population size (Bjorndal et al. 2005). An important caveat for population trends analysis based on nesting beach data is that this may reflect trends in adult nesting females, but it may not reflect overall population growth rates well. Adult nesting females often account for less than 1% of total population numbers. The global abundance of nesting female loggerhead turtles is estimated at 43,320–44,560 (Spotila 2004a).

Pacific Ocean. Abundance has declined dramatically over the past 10-20 years, although loggerheads range widely from Alaska to Chile (NMFS and USFWS 1998d). Pacific nesting is limited to two major locations, Australia and Japan, although low level nesting may occur outside of Japan in areas surrounding the South China Sea (Chan et al. 2007b; Kamezaki et al. 2003). Eastern Australia supported one of the major global loggerhead nesting assemblages until recently (Limpus 1985). Now, less than 500 females nest annually, an 86% reduction in the size of the annual nesting population in 23 years (Limpus and Limpus 2003). The status of loggerhead nesting colonies in southern Japan and the surrounding region is uncertain, but approximately 1,000 female loggerhead turtles may nest there; a 50-90% decline compared to historical estimates (Bolten et al. 1996; Dodd Jr. 1988; Kamezaki et al. 2003; STAJ 2002). Nine major nesting beaches (greater than 100 nests per season) and six "submajor" beaches (10–100 nests per season) exist, including Yakushima island where 40% of nesting occurs (Kamezaki et al. 2003). Nesting declined from an initial peak of approximately 6,638 nests in 1990–1991, followed by a steep decline to a low of 2,064 nests in 1997 (Conant et al. 2009). During the past decade, nesting increased gradually to 5,167 nests in 2005 (Conant et al. 2009), declined and then rose again to a record high of 11,082 nests in 2008, and then 7,495 and 10,121 nests in 2009 and 2010, respectively (STAJ 2008, 2009, 2010).

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

Natural threats. Sea turtles face predation primarily by sharks and to a lesser extent by killer whales. All sea turtles except leatherbacks can undergo “cold stunning” if water temperatures drop below a threshold level, which can pose lethal effects. Eggs are commonly eaten by raccoons and ghost crabs along the eastern U.S. (Barton and Roth 2008). In the water, hatchlings are hunted by herons, gulls, dogfish, and sharks. Heavy loads of barnacles are associated with unhealthy or dead stranded loggerheads (Deem et al. 2009).

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

The major factors inhibiting their recovery include mortalities caused by fishery interactions and degradation of the beaches on which they nest. Shrimp trawl fisheries account for the highest number of captured and killed loggerhead sea turtles. Each year, various fisheries capture loggerhead sea turtles. Along Baja California, it is estimated that 1,500-2,950 loggerheads are killed annually by local fishing fleets (Peckham et al. 2008). Offshore longline tuna and swordfish longline fisheries are also a serious concern for the survival and recovery of loggerhead sea turtles and appear to affect the largest individuals more than younger age classes (Aguilar et al. 1995; Bolten et al. 1994; Carruthers et al. 2009; Howell et al. 2008; Marshall et al. 2009; Petersen et al. 2009; Tomás et al. 2008). Deliberate hunting of loggerheads for their meat, shells, and eggs has declined from previous exploitation levels, but still exists and hampers recovery efforts (Lino et al. 2010). In the Pacific, loggerhead turtles are captured, injured, or killed in numerous Pacific fisheries including

- Japanese longline fisheries in the western Pacific Ocean and South China Seas
- direct harvest and commercial fisheries off Baja California, Mexico
- commercial and artisanal swordfish fisheries off Chile, Columbia, Ecuador, and Peru

- purse seine fisheries for tuna in the eastern tropical Pacific Ocean
- California/Oregon drift gillnet fisheries (NMFS 2006e)

Wallace et al. (2010) estimated that between 1990 and 2008, at least 85,000 sea turtles were captured as bycatch in fisheries worldwide. This estimate is likely at least two orders of magnitude low, resulting in a likely bycatch of nearly half a million sea turtles annually (Wallace et al. 2010); many of these are expected to be loggerhead sea turtles.

Marine debris ingestion can be a widespread issue for loggerhead sea turtles. More than one-third of loggerheads found stranded or bycaught had injected marine debris in a Mediterranean study, with possible mortality resulting in some cases (Lazar and Gračan 2010).

In the Pacific Ocean, between 2,600 and 6,000 loggerhead sea turtles are estimated to have been captured and killed in longline fisheries in 2000 (Lewison et al. 2004b). Shallow-set Hawaii based longline fisheries likely captured and killed several hundred loggerhead sea turtles before their closure in 2001. Upon being re-opened in 2004, with substantial modifications to protect sea turtles, fewer than five loggerhead sea turtles likely experienced fisheries interaction each year. Between 2004 and 2008, shallow-set fisheries based out of Hawaii likely captured about 45 loggerhead sea turtles, killing about 10. Loggerhead sea turtles have also been and are expected to continue to be captured and killed in the deep-set based longline fisheries based out of Hawaii and American Samoa. Although the number of longline tuna vessels (a major component of fisheries in the region) has decreased by over half from 1970 to 2006 and a gill net moratorium is in place for the region, gill nets and longlines are a particular problem in the region, particularly for loggerhead sea turtles (Bowen et al. 1995; Kaplan 2005; Lawson 2008; Lewison et al. 2004b; Polovina et al. 2000; Yokota et al. 2006). An estimated 30,000-75,000 loggerheads were captured as longline bycatch in the Pacific during 2000 (Lewison et al. 2004b). Prior to the gill net moratorium, 16,000 sea turtles were bycaught in the western Pacific and 4,000 loggerheads in the North Pacific (Bowen et al. 1995).

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

Tissues taken from loggerheads sometimes contain very high levels of organochlorines chlorobiphenyl, chlordanes, lindane, endrin, endosulfan, dieldrin, PFOS, PFOA, DDT, and PCB (Alava et al. 2006; Corsolini et al. 2000; Gardner et al. 2003; Keller et al. 2005; Keller et al. 2004a; Keller et al. 2004b; Mckenzie et al. 1999; Monagas et al. 2008; Oros et al. 2009; Perugini et al. 2006; Rybitski et al. 1995; Storelli et al. 2007). It appears that levels of organochlorines have the potential to suppress the immune system of loggerhead sea turtles and may affect metabolic regulation (Keller et al. 2004c; Keller et al. 2006; Oros et al. 2009). These contaminants could cause deficiencies in endocrine, developmental, and reproductive health (Storelli et al. 2007). It is likely that the omnivorous nature of loggerheads makes them more prone to bioaccumulating toxins than other sea turtle species (Godley et al. 1999; Mckenzie et al. 1999).

Heavy metals, including arsenic, barium, cadmium, chromium, iron, lead, nickel, selenium, silver, copper, zinc, and manganese, have also been found in a variety of tissues in levels that increase with turtle size (Anan et al. 2001; Fujihara et al. 2003; Garcia-Fernandez et al. 2009; Gardner et al. 2006; Godley et al. 1999; Saeki et al. 2000; Storelli et al. 2008). These metals likely originate from plants and seem to have high transfer coefficients (Anan et al. 2001; Celik et al. 2006; Talavera-Saenz et al. 2007).

Loggerhead sea turtles have higher mercury levels than any other sea turtle studied, but concentrations are an order of magnitude less than in many toothed whales (Godley et al. 1999; Pugh and Becker 2001). Arsenic occurs at levels several fold more concentrated in loggerhead sea turtles than marine mammals or seabirds.

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

Critical habitat. The NMFS has not designated critical habitat for loggerhead sea turtles.

Olive ridley sea turtle

Distribution. Olive ridleys are globally distributed in tropical regions (>20º C) of the Pacific (southern California to Peru, and rarely in the Gulf of Alaska Hodge and Wing 2000), Indian (eastern Africa and the Bay of Bengal), and Atlantic oceans (Grand Banks to Uruguay and Mauritania to South Africa Foley et al. 2003; Fretey 1999; Fretey et al. 2005; Stokes and Epperly 2006). Olive ridleys are uncommon in the western Pacific and western Indian Oceans, and most of the North Atlantic (Spotila 2004a).

Population designations. Population designations are poorly known. Populations likely correspond somewhat to nesting beach location (Tables 9 and 10). Most olive ridleys nest synchronously in huge events called “arribadas”, with hundreds to thousands of females nesting over the course of three to seven days; other individuals nest alone, out of sequence with the arribada (Aprill 1994b; Kalb and Owens 1994).

Atlantic Ocean. Olive ridley distribution in the western North Atlantic occurs mostly along the northern coast of South America and adjacent waters. In the Caribbean, non-nesting individuals occur regularly near Isla Margarita, Trinidad, and Curacao, but are rare further west, such as in Puerto Rico, the Dominican Republic, and Cuba. In rare cases, olive ridleys are known to occur as far north as Puerto Rico, the Dominican Republic, and Cuba and as far south as Brazil (Moncada-G. 2000 as cited in NMFS 2004b). Regular nesting occurs only in Guyana, Suriname, and French Guiana, with most foraging grounds likely nearby (Reichart 1989 as cited in LGL Ltd. 2007). Nesting occurs along the north coast of Venezuela (Sternberg 1981). Olive ridleys likely occur in low numbers along western Africa.

Pacific Ocean. Typical distribution is from Peru to California, with rare Alaskan sightings. Peak arribada nesting in the eastern Pacific occurs at several beaches in Mexico, Nicaragua, Costa Rica, and Panama (NMFS and USFWS 2007f). Tagged Costa Rican nesters have been recovered as far south as Peru, as far north as Oaxaca, Mexico, and offshore to a distance of 2,000 km. Olive ridleys are the most common sea turtle in oceanic waters of the eastern tropical Pacific but move into nearshore waters prior to breeding (Pitman 1990). This species frequently basks at the surface, is accompanied by seabirds, and associates with floating debris, from logs to plastic debris to dead whales (Arenas and Hall 1991a; Pitman 1992 as cited in NMFS 2004b).

Table 9. Recent estimates of olive ridley arribada size.

Country	Beach	Estimates of arribada size from one-time, most recent counts	References
Western Atlantic Ocean			
Suriname	Galibi Nature Reserve*	335 nests	(Hoekert et al. 1996)
French Guiana		1,716-3,257 females	(Kelle et al. 2009)
Eastern Pacific Ocean			
Nicaragua	Chacocente	42,541 nests	(López Carcache et al. in press)
Nicaragua	La Flor	1,300-9,000 turtles per arribada	(Ruiz 1994)
Nicaragua	Masachapa	No estimate available	(Cornelius 1982; Margaritoulis and Demetropoulos 2003)
Nicaragua	Pochomil	No estimate available	(Cornelius 1982; Margaritoulis and Demetropoulos 2003)
Nicaragua	Boquita	No estimate available	(Cornelius 1982)***
Costa Rica	Nancite	200-20,000 turtles per arribada	(Fonseca et al. 2009)
Costa Rica	Ostional	Average 50,000-200,000 turtles per arribada	(Chaves et al. 2005)
Panama	Isla Cañas	5,000-12,000 turtles per arribada	(Evans and Vargas 1998)
Northern Indian Ocean			
India	Gahirmatha	1,000-100,000+ turtles per arribada	(Shanker et al. 2003)
India	Devi River	No estimate available	(Shanker et al. 2003)
India	Rushikulya	10,000-200,000 turtles per arribada	(Shanker et al. 2003)

* Large arribadas once occurred at these beaches but no longer do (Cliffton et al. 1982; Hoekert et al. 1996).

** These data represent total nests for season.

*** Masachapa, Pochomil, and Boquita were extant at the time of the Cornelius (1982) article. The status for Boquita is unknown.

Table 10. Locations of olive ridley arribada and solitary nesting beaches in the eastern Pacific and estimates of arribada sizes.

Country	Beach	Estimates of arribada size from one-time, most recent counts	References
Arribada			
Mexico	Mismaloya*	1,000-5,000 nests	(R. Briseño and A. Abreu, pers. comm. in NMFS and USFWS 2007c)
Mexico	Tlacoyunque*	500-1,000 nests	(R. Briseño and A. Abreu, pers. comm. in NMFS and USFWS 2007c)
Mexico	Chacahua*	10,000-100,000 nests	(R. Briseño and A. Abreu, pers. comm. in NMFS and USFWS 2007c)
Mexico	La Escobilla	1,000,000+ nests	(Márquez et al. 2005)
Mexico	Moro Ayuta*	10,000-100,000 nests	(R. Briseño and A. Abreu, pers. comm. in NMFS and USFWS 2007c)
Solitary			
Mexico	Entire Pacific coast		(R. Briseño and A. Abreu, pers. comm. in NMFS and USFWS 2007c)

* Large arribadas once occurred at these beaches but no longer do (Cliffton et al. 1982; Hoekert et al. 1996).

Olive ridley sea turtles have rarely been documented along Japan, although they may utilize offshore waters in the region up to the Kuroshio Extension during foraging and developmental migrations (Kamezaki and Matsui 1997; Uchida 1994; Uchida and Nishiwaki 1995). Olive ridleys appear to be more common in the North Pacific Subtropical Gyre, but may forage along the Kuroshio Extension (Polovina et al. 2006; Polovina et al. 2004; Polovina et al. 2003a). Olive ridleys are not known to have been bycaught in the Shatsky Rise region (Yokota et al. 2006).

Southern Hemisphere. Distribution is poorly known, but nesting colonies occur in the Philippines, Papua New Guinea, and northern Australia (Euroturtle 2009; Spring 1982). Solitary nesting beaches occur in Australia, Brunei, Malaysia, Indonesia, and Vietnam (Spotila 2004b). Olive ridleys have been sighted in Fiji, Vanuatu, French Polynesia, the Solomon and Marshall islands, and Palau (SPREP 2007). The occurrence of olive ridleys in Tonga and Kiribati is suspected but unconfirmed (SPREP 2007).

Reproduction and growth. Little is known about olive ridley growth or reproduction. However, some beaches, such as Ostional Beach on the Pacific coast of Costa Rica, are known to have extremely low hatching success, particularly at the onset of the dry season onward, at least partly due to the high temperatures of nests (Valverde et al. 2010).

Migration and movement. Olive ridleys are highly migratory and may spend most of their non-breeding life cycle in deep-ocean waters, but occupy the continental shelf region during the breeding season (Arenas and Hall 1991b; Beavers and Cassano 1996; Cornelius and Robinson

1986; Pitman 1991; Pitman 1993; Plotkin 1994; Plotkin et al. 1994a; Plotkin et al. 1995). Reproductively active males and females migrate toward the coast and aggregate at nearshore breeding grounds near nesting beaches (Cornelius 1986; Hughes and Richard 1974; Kalb et al. 1995; Plotkin et al. 1991; Plotkin et al. 1996; Plotkin et al. 1997; Pritchard 1969). Other males and females may not migrate to nearshore breeding aggregations at all (Kopitsky et al. 2000; Pitman 1991). Some males appear to remain in oceanic waters, are non-aggregated, and mate opportunistically as they intercept females *en route* to near shore breeding grounds and nesting beaches (Kopitsky et al. 2000; Plotkin 1994; Plotkin et al. 1994b; Plotkin et al. 1996). Their migratory pathways vary annually (Plotkin 1994), there is no spatial and temporal overlap in migratory pathways among groups or cohorts of turtles (Plotkin et al. 1994a; Plotkin et al. 1995), and no apparent migration corridors exist. Olive ridleys may use water temperature more than any other environmental cue during migrations (Spotila 2004a). Post-nesting migration routes from Costa Rica traverse more than 3,000 km out into the central Pacific (Plotkin et al. 1994a). Olive ridleys from different populations may occupy different oceanic habitats (Polovina et al. 2004; Polovina et al. 2003b). Unlike other marine turtles that migrate from a breeding ground to a single feeding area, where they reside until the next breeding season, olive ridleys are nomadic migrants that swim hundreds to thousands of kilometers over vast oceanic areas (Plotkin 1994; Plotkin et al. 1994a; Plotkin et al. 1995). Olive ridleys may associate with flotsam, which could provide food, shelter, and/or orientation cues (Arenas and Hall 1991b).

Feeding. Olive ridleys typically forage offshore and feed on a variety of benthic and pelagic species, such as jellyfish, squid, salps, red crabs, acorn and gooseneck barnacles, mollusks, and algae (Márquez 1990; Deraniyagala 1939, Carr 1961, Caldwell 1969, Fritts 1981, Cornelius and Robinson 1986, Mortimer 1982 - as cited in NMFS 2004b).

Diving. Diving behavior remains somewhat of a mystery, but several studies have highlighted general insights. The average dive length for an adult female and male were reported to be 54.3 and 28.5 min, respectively (Plotkin 1994 in Lutcavage and Lutz 1997, as cited in NMFS and USFWS 2007f). McMahon et al. (2007) reported a maximum dive duration of 200 min (± 20 min) in northern Australia. In the eastern tropical Pacific, diving rate is greater during daytime than at night (Beavers and Cassano 1996; Parker et al. 2003). During nighttime however, dives are longer (up to 95 min).

Olive ridleys can dive and feed at considerable depths (80–300 m), although ~90% of their time is spent at depths <100 m (Polovina et al. 2003b). At least 25% of their total dive time is spent in the permanent thermocline, located at 20–100 m (Parker et al. 2003). In the North Pacific Ocean, two olive ridleys tagged with satellite-linked depth recorders spent about 20% of their time in the top meter and about 10% of their time deeper than 100 m; 70% of the dives were no deeper than 5 m (Polovina et al. 2003b).

Vocalization and hearing. Information on the hearing capabilities of sea turtles is limited, but available information supports low-frequency hearing centered below 1 kHz and a hearing threshold at 132-140 dB (Lenhardt 1994; Lenhardt et al. 1983; Moein Bartol and Ketten 2006; Moein Bartol et al. 1999; O'Hara and Wilcox 1990; Ridgway et al. 1969).

Status and trends. Except for the Mexico breeding stock, olive ridley sea turtles were listed as threatened under the ESA on July 28, 1978 (43 FR 32800). The olive ridley is the most abundant sea turtle in the world (Pritchard 1997). Worldwide, abundance of nesting female olive ridleys is estimated at two million (Spotila 2004a).

Atlantic Ocean. Nesting centers, such as around Surinam, have declined more than 80% since 1967. However, nesting along Brazil, Nicaragua, and Costa Rica appear to be increasing, although long-term data are lacking (NMFS and USFWS 2007e).

Pacific Ocean. The eastern Pacific population is believed to number roughly 1.39 million (Eguchi et al. in preperation). Abundance estimates in recent years indicate that the Mismaloya and Moro Ayuta nesting populations appear to be stable and the nesting population at La Escobilla is increasing, although less than historical levels, which was roughly 10 million adults prior to 1950 (Cliffton et al. 1982; NMFS and USFWS 2007c). By 1969, after years of adult harvest, the estimate was just over one million (Cliffton et al. 1982). Olive ridley nesting at La Escobilla rebounded from approximately 50,000 nests in 1988 to over 700,000 nests in 1994, and more than a million nests by 2000 (Márquez et al. 2005; Márquez et al. 1996). The largest known arribadas in the eastern Pacific are on the coast of Costa Rica (~475,000-650,000 females estimated nesting annually) and in southern Mexico (~800,000 nests per year at La Escobilla, in Oaxaca, Mexico). Along Costa Rica, 25,000-50,000 olive ridleys nest at Playa Nancite and 450,000-600,000 turtles nest at Playa Ostional annually (NMFS and USFWS 1998b). At a nesting site in Costa Rica, an estimated 0.2% of 11.5 million eggs laid during a single arribada produced hatchlings (NMFS and USFWS 1998b). Two of the five arribada beaches in Nicaragua have available estimates – Chacocente at over 42,000 nests and La Flor at 1,300 to 9,000 turtles per arribada (NMFS 2004b; NMFS 2004c). Analysis of bycatch data off Costa Rica suggest a female-biased sex ratio of roughly two females for every male (Arauz 2001).

Indian Ocean. Arribada nesting populations are still large but are either in or near decline. Solitary nesting declines have been reported from Bangladesh, Myanmar, Malaysia, Pakistan, and southwest India (NMFS and USFWS 2007e). However, solitary nesting in Indonesia may be increasing (Asrar 1999; Dermawan 2002; Islam 2002; Krishna 2005; Limpus 1995; Thorbjarnarson et al. 2000).

Natural threats. Sea turtles face predation primarily by sharks and to a lesser extent by killer whales. Natural predators of olive ridleys also include crabs, garrabos, iguanas, crocodiles, black vultures, coyotes, raccoons, and coatis (Aprill 1994a). All sea turtles except leatherbacks can undergo “cold stunning” if water temperatures drop below a threshold level, which can pose lethal effects.

Anthropogenic threats. Collection of eggs as well as adult turtles has historically led to species decline (NMFS and USFWS 2007c). Harvests remain a concern for olive ridley recovery. In some locations, takes are now regulated or banned (with varying compliance), while harvests remain uncontrolled in other areas. Adult harvests are now largely banned, except along African coasts.

High levels of adult mortality due to harvesting are believed to be the reason why rapid and large nesting population declines occurred in Mexico (Cornelius et al. 2007). The nationwide ban on commercial sea turtle harvest in Mexico, enacted in 1990, has greatly aided olive ridley conservation, but the population is still seriously decremented and threatened with extinction (Groombridge 1982). Several solitary and arribada nesting beaches experience (although banned) egg harvesting, which is causing declines (Cornelius et al. 2007). Approximately 300,000-600,000 eggs were seized each year from 1995-1998 (Trinidad and Wilson 2000).

In India, uncontrolled mechanized fishing in areas of high sea turtle concentration, primarily illegally operated trawl fisheries, has resulted in large-scale mortality of adult olive ridley turtles during the last two decades. Since 1993, more than 50,000 olive ridleys have stranded along the coast, at least partially because of near-shore shrimp fishing (Shanker and Mohanty 1999). In 2008, several hundred olive ridleys stranded dead along Orissa beaches coincident with trawl fisheries operating in the area (Das 2008). Fishing in coastal waters off Gahirmatha was restricted in 1993 and completely banned in 1997 with the formation of a marine sanctuary around the rookery. However, mortality due to shrimp trawling reached a record high of 13,575 ridleys during the 1997 to 1998 season and none of the approximately 3,000 trawlers operating off the Orissa coast use turtle excluder devices in their nets despite mandatory requirements passed in 1997 (Pandav and Choudhury 1999). Shrimp trawls off of Central America are estimated to capture over 60,000 sea turtles annually, most of which are olive ridleys (Arauz 1996 as cited in NMFS and USFWS 2007f). Olive ridleys in the eastern Pacific are also incidentally caught by purse seine fisheries and gillnet fisheries (Frazier et al. 2007). Wallace et al. (2010) estimated that between 1990 and 2008, at least 85,000 sea turtles were captured as bycatch in fisheries worldwide. This estimate is likely at least two orders of magnitude low, resulting in a likely bycatch of nearly half a million sea turtles annually (Wallace et al. 2010); many of these turtles are expected to be olive ridley sea turtles.

There are additional impacts to the nesting and marine environment that affect olive ridleys. Structural impacts to nesting habitat include the construction of buildings and pilings, beach armoring and renourishment, and sand extraction (Bouchard et al. 1998; Lutcavage et al. 1997). The presence of lights on or adjacent to nesting beaches alters the behavior of nesting adults and is often fatal to emerging hatchlings as they are attracted to light sources and drawn away from the water, with up to 50% of some olive ridley hatchlings disoriented upon emergence in some years (Karnad et al. 2009; Witherington 1992; Witherington and Bjorndal 1991). At sea, there are numerous potential threats including marine pollution, oil and gas exploration, lost and discarded fishing gear, changes in prey abundance and distribution due to commercial fishing, habitat alteration and destruction caused by fishing gear and practices, agricultural runoff, and sewage discharge (Frazier et al. 2007; Lutcavage et al. 1997).

Olive ridley tissues have been found to contain the organochlorines chlordanes, lindane, endrin, endosulfan, dieldrin, DDT, and PCB (Gardner et al. 2003). These contaminants have the potential to cause deficiencies in endocrine, developmental, and reproductive health (Storelli et al. 2007), and are known to depress immune function in loggerhead sea turtles (Keller et al. 2006). Heavy metals, including cadmium, iron, nickel, copper, zinc, and manganese, have been found in a variety of tissues in levels that increase with turtle size (Gardner et al. 2006). Females from sexual maturity through reproductive life should have lower levels of contaminants than males because females offload contaminants to their eggs. Newly emerged hatchlings have higher concentrations than are present when laid, suggesting that metals may be accumulated during incubation from surrounding sands (Sahoo et al. 1996).

Critical habitat. The NMFS has not designated critical habitat for olive ridley sea turtles.

Environmental baseline

By regulation, environmental baselines for 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 that are contemporaneous with the consultation in process (50 CFR §402.02). The *Environmental baseline* for this Opinion includes the effects of several activities affecting the survival and recovery of ESA-listed whale and sea turtle species in the action area.

Physical and oceanographic features

The Shatsky Rise is the largest plateau in the Pacific Ocean, covering three-quarters of a million square kilometers (Sliter and Brown 1993). Three seamounts are found on the plateau.

The region is dominated by two currents, the oligotrophic Kuroshio moving north along Japan's east coast and deflecting (Kuroshio Extension) north and east into the open northwestern Pacific, and the nutrient-rich Oyashio Current, which moves south before meeting the Kuroshio (Aruga et al. 1968; DoN 1994; Kasai et al. 1997; Limsakul et al. 2002; Taniguchi 1999). When these currents meet (roughly between 37-50˚ N and east of 160˚ E), they produce significant upwelling, frontal boundaries, and eddy features that can provide a foundation for local productivity (DoN 1994; Longhurst 2007; Qiu 2001; Qu et al. 2001). In fact, this confluence is one of the most productive regions in the world's oceans (Simard 1995). The primary productivity of the Kuroshio Current has been measured at 402 $\mathrm{mgC\cdot m^{-2}\cdot day^{-1}}$, while the Oyashio is 697 $\mathrm{mgC\cdot m^{-2}\cdot day^{-1}}$ (Sea Around Us 2009). Phytoplankton within the Kuroshio are primarily pico- and nanoplankton (Furuya 1990; Furuya and Marumo 1983). Primary and secondary productivity with the Oyashio is seasonal, with a bloom in late spring decreasing to minimum chlorophyll levels in August before experiencing a secondary bloom in October (Kasai et al. 1997; Limsakul et al. 2002; Odate and Maita 1989; Shiomoto et al. 1994). Waters over the Shatsky Rise experience bloom events in April and May (Komatsu et al. 2002). Secondary productivity is dominated by copepods (Toda 1989; Uye et al. 1996). Smaller copepod varieties are found in the Kuroshio while larger varieties reside in the Oyashio (Ayukai and Hattori 1992; Kobari and Ikeda 1999; Mackas and Tsuda 1999; Tsuda et al. 1999; Tsuda et al. 2001; Tsuda and Sugisaki 1994). In addition, the maximum concentrations of zooplankters in each of these systems is different (23-54 mg $\mathrm{m^{-3}}$ for the Kuroshio versus 22-183 mg $\mathrm{m^{-3}}$ for the Oyashio) (Limsakul et al. 2002).

Climate change

We primarily discuss climate change as a threat common to all species addressed in this Opinion, rather than in each of the species-specific narratives. As we better understand responses to climate change, we will address these effects in the relevant species-specific section.

In general, based on forecasts made by the Intergovernmental Panel on Climate Change, climate change is projected to have substantial direct and indirect effects on individuals, populations, species, and the structure and function of marine, coastal, and terrestrial ecosystems in the near future (IPCC 2000; IPCC 2001a; IPCC 2001b; IPCC 2002). From 1906 to 2006, global surface temperatures have risen 0.74º C and continues at an accelerating pace; 11 of the 12 warmest years on record since 1850 have occurred since 1995 (Poloczanska et al. 2009). Furthermore, the Northern Hemisphere (where a greater proportion of ESA-listed species occur) is warming faster than the Southern Hemisphere, although land temperatures are rising more rapidly than over the oceans (Poloczanska et al. 2009). The direct effects of climate change will result in increases in atmospheric temperatures, changes in sea surface temperatures, patterns of precipitation, and sea level. Oceanographic models project a weakening of the thermohaline circulation resulting in a reduction of heat transport into high latitudes of Europe, an increase in the mass of the Antarctic ice sheet, and a decrease in the Greenland ice sheet, although the magnitude of these changes remain unknown. Species that are shorter-lived, larger body size, or generalist in nature are liable to be better able to adapt to climate change over the long term versus those that are longer-lived, smaller-sized, or rely upon specialized habitats (Brashares 2003; Cardillo 2003; Cardillo et al. 2005; Issac 2009; Purvis et al. 2000). Climate change is most likely to have its most pronounced affects on species whose populations are already in tenuous positions (Isaac 2008). As such, we expect the risk of extinction to listed species to rise with the degree of climate shift associated with global warming.

The indirect effects of climate change would result from changes in the distribution of temperatures suitable for whale calving and rearing, the distribution and abundance of prey and abundance of competitors or predators. For species that undergo long migrations, individual movements are usually associated to prey availability or habitat suitability. If either is disrupted by changing ocean temperature regimes, the timing of migration can change or negatively impact population sustainability (Simmonds and Eliott. 2009). Climate change can influence reproductive success by altering prey availability, as evidenced by low-success of northern elephant seals during El Niño periods, when cooler, more productive waters are associated with higher first year pup survival (McMahon and Burton. 2005). Reduced prey availability resulting from increased sea temperatures has also been suggested to explain reductions in Antarctic fur seal pup and harbor porpoise survival (Forcada et al. 2005; Macleod et al. 2007). Polygamous marine mammal mating systems can also be perturbated by rainfall levels, with the most competitive grey seal males being more successful in wetter years than in drier ones (Twiss et al. 2007). Sperm whale females were observed to have lower rates of conception following unusually warm sea surface temperature periods (Whitehead 1997). Marine mammals with restricted distributions linked to water temperature may be particularly exposed to range restriction (Issac 2009; Learmonth et al. 2006). MacLeod (2009) estimated that, based upon expected shifts in water temperature, 88% of cetaceans would be affected by climate change, 47% would be negatively affected, and 21% would be put at risk of extinction. Of greatest concern are cetaceans with ranges limited to non-tropical waters and preferences for shelf habitats (Macleod 2009). Kaschner et al. (2011)modeled marine mammal species richness, overlaid with projections of climate change and found that species in lower-latitude areas would likely be more affected than those in higher-latitude regions. Variations in the recruitment of krill and the reproductive success of krill predators correlate to variations in sea-surface temperatures and the extent of sea-ice cover age during winter months. Although the IPCC (2001b) did not detect significant changes in the extent of Antarctic sea-ice using satellite measurements, Curran et al. (2003) analyzed ice-core samples from 1841 to 1995 and concluded Antarctic sea ice cover had declined by about 20% since the 1950s.

Roughly 50% of the Earth’s marine mammal biomass occurs in the Southern Ocean, with all baleen whales feeding largely on a single krill species, *Euphausia superba*, here and feeding virtually nowhere else(Boyd 2002). However, Atkinson et al. (2004) found severe decreases in krill populations over the past several decades in some areas of the Antarctic, linked to sea ice loss. Reid and Croxall (2001) analyzed a 23-year time series of the reproductive performance of predators (Antarctic fur seals, gentoo penguins, macaroni penguins, and black-browed albatrosses) that depend on krill for prey and concluded that these populations experienced increases in the 1980s followed by significant declines in the 1990s accompanied by an increase in the frequency of years with reduced reproductive success. The authors concluded that macaroni penguins and black-browed albatrosses had declined by as much as 50% in the 1990s, although incidental mortalities from longline fisheries probably contributed to the decline of the albatross. However, these declines resulted, at least in part, from changes in the structure of the krill population, particularly reduced recruitment into older krill age classes, which lowered the number of predators krill could sustain. The authors concluded that the biomass of krill within the largest size class was sufficient to support predator demand in the 1980s but not in the 1990s. By 2055, severe reductions in fisheries catch due to climate change have been suggested to occur in the Indo-Pacific, Red Sea, Mediterranean Sea, Antarctic, and tropical areas worldwide while increased catches are expected in the Arctic, North Pacific, North Atlantic, and northern portions of the Southern Ocean (Cheung et al. 2010).

Similarly, Sims et al. (2001) found the timing of squid peak abundance in the English Channel advanced by 120-150 days in the warmest years compared with the coldest. Bottom water temperatures correlated with the extent of squid movement, and temperature increases over the 5 months before and during the month of peak squid movement did not differ between early and late years. These authors concluded that the temporal variation in peak abundance of squid seen off Plymouth represents temperature-dependent movement, which climatic changes association with the North Atlantic Oscillation mediate. Cephalopods dominate the diet of sperm whales, who would likely re-distribute following changes in the distribution and abundance of their prey. If, however, cephalopod populations collapse or decline dramatically, sperm whales would likely decline as well.

Climate change has been linked to changing ocean currents as well. Rising carbon dioxide levels have been identified as a reason for a poleward shift in the Eastern Australian Current, shifting warm waters into the Tasman Sea and altering biotic features of the area (Johnson et al. 2011; Poloczanska et al. 2009). Similarly, the Kuroshio Current in the western North Pacific (an important foraging area for juvenile sea turtles) has shifted southward as a result of altered long-term wind patterns over the Pacific Ocean (Poloczanska et al. 2009).

Climate-mediated changes in the distribution and abundance of keystone prey species like krill and climate-mediated changes in the distribution of cephalopod populations worldwide is likely to affect marine mammal populations as they re-distribute throughout the world's oceans in search of prey. If sea ice extent decreases, then larval krill may not be able to survive without access to underice algae to feed on. This may be a cause of decreased krill abundance in the northern western Antarctic Peninsula during the last decade (Fraser and Hofmann 2003). Meltwaters have also reduced surface water salinities, shifting primary production along the Antarctic Peninsula (Moline et al. 2004). Blue whales, as predators that specialize in eating krill, are likely to change their distribution in response to changes in the distribution of krill (Clapham et al. 1999; Payne et al. 1986; Payne et al. 1990b). If they did not change their distribution or could not find the biomass of krill necessary to sustain their population numbers, their populations would likely experience declines similar to those observed in other krill predators, including dramatic declines in population size and increased year-to year variation in population size and demographics. These outcomes would dramatically increase the extinction probability of baleen whales. Edwards et al. (2007) found a 70% decrease in one zooplankton species in the North Sea and an overall reduction in plankton biomass as warm-water species invade formerly cold-water areas. However, in other areas, productivity may increase, providing more resources for local species (Brown et al. 2009). In addition, reductions in sea ice may alleviate "choke points" that allow some marine mammals to exploit additional habitats (Higdon and Ferguson 2009).

Foraging is not the only potential aspect that climate change could influence. Acevedo-Whitehouse and Duffus (2009) proposed that the rapidity of environmental changes, such as those resulting from global warming, can harm immunocompetence and reproductive parameters in wildlife to the detriment of population viability and persistence. An example of this is the altered sex ratios observed in sea turtle populations worldwide (Fuentes et al. 2009a; Mazaris et al. 2008; Reina et al. 2008; Robinson et al. 2008). This does not appear to have yet affected population viabilities through reduced reproductive success, although nesting and emergence dates of days to weeks in some locations have changed over the past several decades (Poloczanska et al. 2009). Altered ranges can also result in the spread of novel diseases to new areas via shifts in host ranges (Simmonds and Eliott. 2009). It has also been suggested that increases in harmful algal blooms could be a result from increases in sea surface temperature (Simmonds and Eliott. 2009).

Changes in global climatic patterns will likely have profound effects on the coastlines of every continent by increasing sea levels and the intensity, if not the frequency, of hurricanes and tropical storms (Wilkinson and Souter 2008). A half degree Celsius increase in temperatures during hurricane season from 1965-2005 correlated with a 40% increase in cyclone activity in the Atlantic. Sea levels have risen an average of 1.7 mm/year over the 20th century due to glacial melting and thermal expansion of ocean water; this rate will likely increase. Based on computer models, these phenomena would inundate nesting beaches of sea turtles, change patterns of coastal erosion and sand accretion that are necessary to maintain those beaches, and would increase the number of turtle nests destroyed by tropical storms and hurricanes (Wilkinson and Souter 2008). The loss of nesting beaches, by itself, would have catastrophic effects on sea turtle populations globally if they are unable to colonize new beaches that form or if the beaches do not provide the habitat attributes (sand depth, temperature regimes, refuge) necessary for egg survival. In some areas, increases in sea level alone may be sufficient to inundate sea turtle nests and reduce hatching success (Caut et al. 2009b). Storms may also cause direct harm to sea turtles, causing “mass” strandings and mortality (Poloczanska et al. 2009). Increasing temperatures in sea turtle nests alters sex ratios, reduces incubation times (producing smaller hatchling), and reduces nesting success due to exceeded thermal tolerances (Fuentes et al. 2009b; Fuentes et al. 2010; Fuentes et al. 2009c). Smaller individuals likely experience increased predation (Fuentes et al. 2009b)Climatic anomalies influencing the Marianas Islands include El Niño/Southern Oscillation (ENSO) and La Niña events (Giese and Carton 1999; Mantua and Hare 2002a; NOAA 2005a; NOAA 2005b; Sugimoto et al. 2001; Trenberth 1997). Although Guam and the Southern Marianas Islands do not appear to experience altered rainfall patterns during El Niño events, the Northern Marianas tend to experience drier dry seasons and wetter wet seasons (Pacific ENSO Applications Center 1995). Sea surface temperature in the regions also increases due to a weakening of a high pressure system over the western Pacific, potentially influencing the distribution of fish (Kubota 1987; Lehodey et al. 1997). Although typhoons tend to be more frequent during El Niño events (likely occurring at present), their tracks tend to be more to the northwest, away from the action area (Elsner and Liu 2003; Saunders et al. 2000).

Unlike El Niño and La Niña events, Pacific Decadal Oscillation events can persist for 20-30 years, but are more prominent outside the tropics, and mechanisms controlling them are relatively unknown (Hare and Mantua 2000; Mantua and Hare 2002b; Minobe 1997; Minobe 1999). PDO events should not strongly influence the action area.

Habitat degradation

A number of factors may be directly or indirectly affecting listed species in the action area by degrading habitat. These include ocean noise and fisheries impacts.

Natural sources of ambient noise include: wind, waves, surf noise, precipitation, thunder, and biological noise from marine mammals, fishes, and crustaceans. Anthropogenic sources of ambient noise include: transportation and shipping traffic, dredging, construction activities, geophysical surveys, and sonars. In general, it has been asserted that ocean background noise levels have doubled every decade for the last six decades in some areas, primarily due to shipping traffic (IWC 2004). The acoustic noise that commercial traffic contributes to the marine environment is a concern for listed species because it may impair communication between individuals (Hatch et al. 2008).

Seamounts are sensitive to fishery impacts due to the high level of endemism characteristic of this habitat. Species that inhabit seamounts tend to be long-lived and do not move widely between seamounts, meaning that their recovery can be very slow (Johnston and Santillo 2004; Richer de Forges 2000). Listed species may associate with seamounts, apparently due to prey availability here, and the deterioration of the habitat can have significant effects on listed species.

Vessel traffic

Vessel noise could affect marine animals in the proposed study area. Shipping and seismic noise generally dominates ambient noise at frequencies from 20 to 300 Hz (Andrew et al. 2002; Hildebrand 2009; Richardson et al. 1995c). Background noise has increased significantly in the past 50 years as a result of increasing vessel traffic, and particularly shipping, with increases of as much as 12 dB in low frequency ranges; background noise may be 20 dB higher now versus preindustrial periods (Hildebrand 2009; Jasny et al. 2005; McDonald et al. 2006; NRC 1994; NRC 2003; NRC 2005; Richardson et al. 1995a). Over the past 50 years, the number of commercial vessels has tripled, carrying an estimated six times as much cargo (requiring larger, more powerful vessels)(Hildebrand 2009). Seismic signals also contribute significantly to the low frequency ambient sound field (Hildebrand 2009). Baleen whales may be more sensitive to sound at those low frequencies than are toothed whales. Dunlop et al. (2010) found that humpback whales shifted from using vocal communication (which carries relatively large amounts of information) to surface-active communication (splashes; carry relatively little information) when low-frequency background noise increased due to increased sea state. Sonars and small vessels also contribute significantly to mid-frequency ranges (Hildebrand 2009).

Scientific and research activities

Scientific research permits issued by the NMFS currently authorize studies on listed species in the Pacific Ocean, which may extend into portions of the action area for the proposed survey. Authorized research on ESA-listed whales includes close vessel and aerial approaches, biopsy sampling, tagging, ultrasound, exposure to acoustic activities, and breath sampling. Research activities involve non-lethal “takes” of these whales by harassment, with none resulting in mortality.

Tables 11-21 describe the cumulative number of takes for each listed species in the action area authorized in scientific research permits. Although most research will take individuals in the Northern Hemisphere, some takes may include individuals from Southern Hemisphere populations.

Table 11. Blue whale takes in the North Pacific.

Year	Approach	Biopsy	Suction cup tagging	Implantable tagging	Acoustic playback	Breath sampling
2009	13,044	1,065	313	105	0	0
2010	15,747	1,475	538	180	21	0
2011	23,677	2,570	3,455	1,295	21	1,020
2012	17,647	2,570	3,435	1,295	21	1,020
2013	12,902	2,460	3,295	1,250	21	1,020
Total	83,017	10,140	11,016	4,125	84	3,060

Permit numbers: 540-1811, 727-1915, 731-1774, 774-1714, 781-1824, 782-1719, 808-1735, 1058-1733, 1071-1770, 1127-1921, 14097, 14122, 14245, 14296, 14451, 14534, 14585, 15271, and 15330.

Table 12. Fin whale takes in the North Pacific.

Year	Approach	Biopsy	Suction cup tagging	Implantable tagging	Acoustic playback	Breath sample
2009	16,860	1,975	150	55	0	0
2010	21,283	2,975	410	130	80	0
2011	35,342	4,664	3,880	209	80	1,020
2012	24,642	4,664	3,880	209	80	1,020
2013	21,367	4,604	3,840	199	80	1,020
Total	119,494	18,882	12,160	802	320	3,060

Permit numbers: 0642-1536, 473-1700, 540-1911, 731-1774, 774-1714, 781-1824, 782-1719, 808-1735, 965-1821, 1049-1718, 1058-1733, 1071-1770, 1127-1921, 14097, 14122, 14296, 14451, and 14534.

Table 13. Humpback whale takes in the North Pacific.

Year	Approach	Biopsy	Suction cup tagging	Implantable tagging	Acoustic playback	Exhalation sampling	Ultrasound
2009	44,399[1]	4,650	392	77	280	10	5
2010	67,171	6,060	1,447	237	970	10	5
2011	85,301	6,700	6,755	1,590	690	1,070	5
2012	50,697	6,500	6,585	1,565	990	1,070	5
2013	37,627	6,035	6,515	1,525	390	1,060	0
Total	285, 195	21,694	21,694	4,994	3,320	3,220	20

Permit numbers: 0642-1536, 0662-1661, 473-1700, 545-1761, 532-1822, 540-1811, 587-1767, 716-1705, 731-1774,753-1599, 727-1915, 774-1714, 781-1824, 782-1719, 808-1753, 945-1776, 965-1821, 1049-1718, 1058-1733, 1071-1770, 1120-1898, 1127-1921, 10018, 14097, 14122, 14245, 15271, 15274, 14296, 14353, 14451, 14534, 14585, 14599, 14610,14682, 13846, and 15330.

Table 14. North Pacific right whale takes in the North Pacific.

Year	Approach/harass	Biopsy	Suction cup tagging	Implantable tagging	Acoustic playback
2009	717	88	42	42	0
2010	902	118	62	52	50
2011	1,302	188	132	62	50
2012	850	138	130	60	50
2013	703	138	94	24	50
Total	4,474	670	460	240	200

Permit numbers: 782-1719, 1058-1733, 13846, 14097, 14122, 14245, 14296, 14585, and 15330.

Table 15. Sei whale takes in the North Pacific.

Year	Approach	Biopsy	Suction cup tagging	Implantable tagging	Exhalation sampling
2009	4,170[1]	435	75	25	0
2010	6,396[1]	730	250	100	0
2011	7,871	638	548	115	1,060
2012	5,551	638	548	115	1,060
2013	5,331	628	558	115	1,060
Total	29,519	3,069	1,969	470	3,180

Permit numbers: 0642-1536, 540-1811, 727-1915, 731-1774, 774-1714, 782-1719, 808-1735, 1049-1718, 1058-1733, 1127-1921, 14097, 14122, 14245, 14296, 14451, 14534, 14585, and 15330.

[1] The National Marine Mammal Laboratory was granted unlimited takes via approach harassment in association with surveys conducted in the North Pacific.

Table 16. Sperm whale takes in the North Pacific Ocean.

Year	Approach	Biopsy	Suction cup tagging	Implantable tagging	Acoustic playback	Exhalation sampling
2009	17,895	770	100	40	0	0
2010	22,001	1,425	405	170	120	120
2011	34,621	3,785	2,885	380	120	1,060
2012	19,486	3,285	2,855	370	120	1,060
2013	18,476	3,165	2,855	360	120	1,060
Total	112,479	12,430	9,100	1,320	480	3,300

Permit numbers: 0642-1536, 473-1700, 540-1811, 727-1915, 731-1774, 774-1714, 781-1824, 782-1719, 1049-1718, 1071-1770,1127-1921, 14097, 14122, 14245, 14296, 14451, 14534, 14585, and 15330.

Table 17. Green sea turtle takes in the North Pacific Ocean.

Year	Capture/handling /restraint	Satellite/sonic/ pit/flipper tagging	Blood/tissue collection	Lavage	Ultrasound	Tetracycline injection
2009	1,066	1,060	766	345	0	85
2010	1,123	1,117	823	380	45	120
2011	1,048	1,042	748	380	45	120
2012	448	442	448	180	45	120
2013	263	257	263	95	45	45
Total	3,711	3,681	3,048	1,380	180	490

Permit numbers: 1514, 1537, 1556, 1581, 1591, 10027, 14097, 14381, and 14510.

Table 18. Hawksbill sea turtle takes in the North Pacific Ocean.

Year	Capture/handling /restraint	Satellite/sonic/ pit/flipper tagging	Blood/tissue collection	Lavage
2009	122	120	122	10
2010	132	130	122	10
2011	102	100	92	10
2012	92	90	82	10
2013	52	50	42	10
Total	500	490	460	50

Permit numbers: 1556, 1581, 10027, 14097, and 14381.

Table 19. Leatherback sea turtle takes in the North Pacific Ocean.

Year	Capture/handling/restraint	Satellite/sonic/ PIT/flipper tag	Blood/tissue collection	Ultrasound
2009	191	191	191	38
2010	222	222	222	38
2011	222	222	222	38
2012	188	188	188	38
2013	112	112	122	0
Total	935	935	935	152

Permit numbers: 1596, 14381, 14097, and 14510.

Table 20. Loggerhead sea turtle takes in the Pacific Ocean.

Year	Approach	Capture/ handling/ restraint	Satellite/ sonic/ flipper tagging	Blood/tissue collection	Lavage	Ultrasound	Tetra-cycline injection
2009	49	49	49	49	18	0	8
2010	111	111	111	111	24	9	14
2011	90	90	90	90	24	9	14
2012	90	90	90	90	24	9	14
2013	82	82	82	82	16	9	6
Total	422	422	422	422	106	36	56

Permit numbers: 1514, 1591, 14097, 14381, and 14510.

Table 21. Olive ridley sea turtle takes in the Pacific Ocean.

Year	Approach	Capture/ handling/ restraint	Satellite/ sonic/ flipper tagging	Blood/tissue collection	Lavage	Ultrasound	Tetra-cycline injection
2009	351	351	351	351	58	0	8
2010	404	404	404	404	64	7	8
2011	361	361	361	361	64	7	8
2012	361	361	361	361	64	7	8
2013	353	353	353	353	56	7	0
Total	1,830	1,830	1,830	1,830	306	28	32

Permit numbers: 1514, 1591, 14097, 14381, and 14510.

Effects of the proposed actions

Pursuant to Section 7(a)(2) of the ESA, federal agencies must insure, through consultation with the NMFS, 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 funding by the NSF of the seismic survey and issuance of the IHA by the 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 to reduce an individual's fitness (i.e., growth, survival, annual reproductive success, or 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 and, ultimately, of the Opinion is to determine if it is reasonable to expect the proposed action 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 action would authorize non-lethal "takes" by harassment of listed species during seismic survey activities. The ESA does not define harassment nor has the NMFS defined the term pursuant to the ESA through regulation. 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"[2] 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.

Our analysis considers that behavioral harassment or disturbance is not limited to the "take" definition and may in fact occur in many ways. Fundamentally, if our analysis leads us to conclude that an individual changes its behavioral state (for example, from resting to traveling away from the airgun source or from traveling to evading), we consider the individual to have been harassed or disturbed, regardless of whether it has been exposed to acoustic criteria that define "take." In addition, individuals may respond in a variety of ways, some of which have more significant fitness consequences than others. For example, evasion of a seismic source would be more significant than slow travel away from the same stressor due to increased metabolic demands, stress responses, and potential for calf abandonment that this response could or would entail. As described in the *Approach to the assessment*, the universe of likely responses is considered in evaluating the fitness consequences to the individual and (if appropriate), the affected population and species as a whole to determine the likelihood of jeopardy.

Potential stressors

The assessment for this consultation identified several possible stressors associated with the proposed seismic activities, including

1. pollution by oil or fuel leakage
2. ship-strikes
4. acoustic interference from engine noise
5. entanglement in towed hydrophone streamer
6. sound fields produced by airguns, sub-bottom profiler, and multibeam echosounder

Based on a review of available information, this Opinion determined which of these possible stressors would be likely to occur and which would be discountable or insignificant. The potential for fuel or oil leakages and ship strikes are extremely unlikely. The former would likely pose a significant risk to the vessel and its crew and actions to correct a leak should occur immediately to the extent possible. In the event that a leak should occur, the amount of fuel and oil onboard the *Langseth* is unlikely to cause widespread, high dose contamination (excluding the remote possibility of severe damage to the vessel) that would impact listed species directly or pose hazards to their food sources.

As stated in *Description of the proposed actions*, the propulsion system of the *Langseth* is very

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[2] 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)

quiet compared to other vessels to reduce interference with seismic activities. Although noise originating from vessel propulsion will propagate into the marine environment, this amount would be so small as to be discountable. The *Langseth*'s passage past a whale or sea turtle would be brief and not likely to be significant in impacting any individual's ability to feed, reproduce, or avoid predators. Brief interruptions in communication via masking are possible, but unlikely given the habits of whales to move away from vessels, either as a result of engine noise, the physical presence of the vessel, or both (Lusseau 2006). In addition, the *Langseth* will be traveling at slow speeds, reducing the amount of noise produced by the propulsion system and the probability of a ship-strike (Kite-Powell et al. 2007; Vanderlaan and Taggart 2007). We are not aware of a ship-strike by a seismic survey vessel. Our expectation of shipstrike is discountably small due to low whale density (except sperm whales) and general expected movement away or parallel to the *Langseth* (Hauser and Holst 2009; Holst 2009; Holst 2010; Holst and Smultea 2008a). All things considered, we feel the potential for ship strike or acoustic interference from propulsion noise is discountable. Towed hydrophone streamer or passive acoustic array could come in direct contact with a listed species and sea turtle entanglements have occurred in towed seismic gear. For example, a seismic survey in the eastern tropical Pacific during 2011 recovered a dead olive ridley sea turtle in the foil of towed seismic gear; it is unclear whether the sea turtle became lodged in the foil pre- or post mortem (Spring 2011). However, entanglement is highly unlikely due to the streamer design as well as observations of sea turtles investigating the streamer and not becoming entangled or operating in regions of high turtle density and entanglements not occurring (Hauser et al. 2008; Holst and Smultea 2008a; Holst et al. 2005a; Holst et al. 2005b). Although the towed hydrophone streamer or passive acoustic array could come in direct contact with a listed species, entanglements are highly unlikely and considered discountable.

Accordingly, this consultation focused on the following stressors likely to occur from the proposed seismic activities and may adversely affect ESA-listed species: 1. acoustic energy introduced into the marine environment by the airgun array and 2. acoustic energy introduced by both the multibeam echosounder and sub-bottom profiler sonars.

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 "harassed," under the MMPA (65 FR 16374). These thresholds help to develop exclusion radii around a source and the necessary power-down or shut-down criteria. Airguns contribute a massive amount of anthropogenic energy to the world's oceans (3.9×10^{13} joules), second only to nuclear explosions (Moore and Angliss 2006). Although most energy is in the low-frequency range, airguns emit a substantial amount of energy up to 150 kHz (Goold and Coates 2006). Seismic airgun noise can propagate substantial distances at low frequencies (e.g., Nieukirk et al. 2004).

The exposure analysis for this Opinion is concerned with the number of blue, sei, fin, humpback, North Pacific right, and sperm whales, as well as green, hawksbill, leatherback, loggerhead, and olive ridley sea turtles likely to be exposed to received levels greater than 160 dB re 1 µPa_{rms} (166 for sea turtles), which constitute the best estimate of adverse response by listed whales and sea turtles. The NSF and Permits and Conservation Division estimated the expected number of ESA-listed whales exposed to received levels ≥160 dB re 1 µPa_{rms}. The data and methodology used were adopted in this Opinion because the Endangered Species Act Interagency Cooperation Division believed they represent the best available information and methods to evaluate exposure to listed species.

The NSF and Permits and Conservation Division provided density estimates for listed whales in the action area. Review of the local survey data and surveys in the wider North Pacific as well as knowledge of listed species life history and local oceanographic conditions supports these estimates as the best available information. Density estimates for cetaceans are based upon data obtained by Japanese whaling surveys as part of JARPN/JARPN II projects. Data summaries were submitted to the IWC in several reports; the IWC has accepted these reports as representative of whales in the northwestern Pacific Ocean. Sei whale density was derived from 2002-2003 surveys of the area between 35-50º N and 150-170º E during August and September, excluding the Russian EEZ (Hakamada et al. 2004). Sperm whale density originate from 1982-1996 survey data in the region of 20-50º N and 130º E-180º (Kato and Miyashita 1998). Density estimates for blue, fin, North Pacific right, and humpback whales stem from surveys in the North Pacific between 31-51º N and 140-170º E from 1994-2007 (Matsuoka et al. 2009). Data for these estimates include three survey areas, one of which was relatively coastal (i.e., habitat unlike the action area while the other two survey areas are similar in habitat and in close proximity to/overlap the action area). The relatively coastal survey area had very few blue and fin whale sightings, but appeared to have encompassed a significant portion of the survey effort. However, we could not determine the amount of survey effort expended in the coastal survey area and therefore could not analyze the remaining survey areas that likely are more representative of blue and fin whale occurrence in the action area independent of the more coastal survey area. Thus, the blue and fin whale density estimates, and therefore exposure estimates, are likely biased by the greater survey effort in the coastal area.

All densities (except sperm whales) are likely overestimates of true density because we expect all other listed whale populations that would be exposed to the proposed seismic survey to still be engaged in migration into the region at the time of the seismic survey. Although some individuals are expected to be present, be expect a sizeable component of each population to be far away from the action area. As we could not determine a method to account for this (survey effort during the time of the proposed seismic survey is fairly low), we accepted a likely inflated estimate of true density so as to avoid underestimating the effects of the action while maintaining an exposure estimate that is reasonably likely to occur.

Marine mammals are expected to be abundant in the action area. The region just north and east of the action area is where two currents, the oligotrophic Kuroshio and the nutrient-rich Oyashio, meet, producing significant upwelling, frontal boundaries, and eddy features that can provide a foundation for local productivity (Aruga et al. 1968; DoN 1994; Kasai et al. 1997; Limsakul et al. 2002; Longhurst 2007; Qiu 2001; Qu et al. 2001; Taniguchi 1999). This confluence results in one of the most productive regions in any ocean (Simard 1995). The high levels of primary and secondary productivity support high densities of listed whales observed during JARPN/JARPN II surveys. Whale densities (excluding sei whales) derived from JARPN/JARPN II surveys are similar to those found along the western U.S., another foraging area for several listed species when present for foraging (Barlow and Forney 2007; Hakamada et al. 2004; Kato and Miyashita 1998; Matsuoka et al. 2009). April and May are bloom periods in the Oyashio Current over the Shatsky Rise (Kasai et al. 1997; Komatsu et al. 2002; Limsakul et al. 2002; Odate and Maita 1989; Shiomoto et al. 1994). Secondary and higher trophic level growth tend to lag behind primary productivity, often by weeks or longer, and prey may still be available to support feeding of baleen whale species.

L-DEO also estimated the exposure radii around the proposed *Langseth* operations using empirical data gathered in the Gulf of Mexico in 2007-2008 aboard the *Langseth*. The distances to which sound levels (rms) might propagate for single airgun and full airgun arrays used during the proposed study are provided in Table 1 on page 6. The maximum distance from airguns where received levels might reach 160 dB re 1 μPa_{rms} (i.e., from the full 36-gun array) at 2,000 m depth (maximum depth at which listed species are expected to occur) is 3,850 m with a 9 m tow depth. A thorough review of available literature (see *Response analysis*) supports this level as a general point at which baleen whales tend to show some avoidance response to received seismic sound. The NSFs assumption that individuals will move away if they experience sound levels high enough to cause significant stress or functional impairment is also reasonable (see *Response analysis*). Isopleth modeling tends to overestimate the distance to which various isopleths will propagate because most exposure will likely occur at depths shallower than 2,000 m, where received sound levels should be reduced. As we are unable to know where individuals will be in the water column at the time of exposure, we accept this assumption. In addition, the 160 dB re 1 μPa_{rms} 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). It should be noted that, although a received level of 166 dB re 1 μPa_{rms} is considered here to be the threshold for harassment for sea turtle response (McCauley et al. 2000a; McCauley et al. 2000b), estimates of this range were not available and the more conservative range at the 160 dB re 1 μPa_{rms} isopleth was used to estimate sea turtle harassment instead, as it was the best estimate available.

A major mitigation factor proposed by the NSF is visual monitoring, especially for marine mammals, which should reduce exposure of listed whales and sea turtles. However, visual monitoring has several limitations. Although regions ensonified by 160 dB re 1 μPa_{rms} and 180 dB re 1 μPa_{rms} are both within the visual range of the *Langseth* and its observers, it is unlikely that all listed species are easily visible at this distance. Ramp-down and shut-down procedures are unlikely to be completely effective at eliminating the co-occurrence of listed individuals within the sound field ≥160 dB re 1 μPa_{rms}.

Our assessment of the proposed seismic survey is unique from prior NSF seismic surveys in that a nearly identical seismic survey was conducted in the same area in 2010, although during a somewhat different time frame (mid July-mid September). The proposed seismic survey is a continuation of this prior survey, as seismic acquisition could not be completed in the allotted time due to emergency medical conditions onboard the *Langseth*. A monitoring report was provided subsequent to this survey that reported sightings of three sperm whale groups (12 individuals) and one unidentified whale over the course of 7,300 km of effort in the action area (Holst and Beland 2010). One of the three sperm whale group sightings (seven individuals) occurred during seismic operations, during which individuals were swimming parallel to the *Langseth* at a distance of 1 km, or just outside of the modeled 180 dB isopleth. In addition, two unidentified sea turtles were sighted in the action area. One ramp-down for sperm whales occurred just outside of the 180 dB isopleth. This survey occurred within a time frame during which we would expect listed marine mammals to be more abundant than during the time frame of the proposed seismic survey, as we expect a lower proportion of individuals within listed whale populations (except sperm whales) to occupy the region during March-May versus June-September. Although exposure to several listed species was not observed during the 2010 cruise, this does not mean that exposure did not occur or cannot occur. We view the monitoring data collected during 2010 as support that low numbers of listed marine mammals and sea turtles can be exposed to the proposed action.

Marine mammals

Exposure of listed mammals to airguns. The exposure estimates stem from the best available information on whale densities and a planned ensonified area of approximately 11,536 km^2 along survey track lines, including areas of repeated exposure.

NSF's exposure estimates (Table 22) were calculated by using the density per 1,000 km^2 multiplied by the total survey track area (11,536 km^2) to obtain the total number of exposures (rounded to the next whole number). The NSF's exposure estimates include repeated exposure of individuals (calculated as above, but excluding the area ensonified multiple times by adjacent tracklines; 11,536 km^2). The NSF assumes that individuals would not move within their environment; an assumption that is highly unlikely. We expect listed individuals to move in their environment to feed on available prey, continue migration, or complete other life functions. There is no known factor by which we can account for the probability of movement, and we do not know of a mechanism by which we can accurately calculate the number of exposures per individual in this situation. Therefore, we accept NSF's approach pending better information, even though we assume NSF's numbers overestimate the likely exposure per individual animal. The number of North Pacific right whale exposures calculated from this methodology was increased from one to two based upon mean group size (Matsuoka et al. 2009).

Whales of all age classes are likely to be exposed. Listed whales are expected to be feeding in the area and some females would have young-of-the-year accompanying them. We would normally assume that sex distribution is even for whales and sexes are exposed at a relatively equal level. However, Tamura et al. (2009) provided estimates of sex and age classes based upon whaling data from the area during March through June for sei whales. Based upon this, we expect three immature males, seven immature females, three adult males, and eight adult females would be exposed. As we lack similar data for any other listed whale, we assume an even sex distribution and cannot estimate exposures to particular age classes.

Sperm whale exposure is somewhat different. Adult sperm whale males tend to feed in higher latitudes. However, this is not the reason for differential exposure for the present survey. Matsuoka et al. (2005) found schools of multiple individuals (family groups, containing all age and sex classes except adult and subadult males, or bachelor schools containing subadult males) were generally found west of the proposed action area, whereas solitary individuals (adult males) were broadly distributed throughout the region. Therefore, exposure is expected to primarily occur to adult males, although any age or sex class could be exposed. As with baleen whales, sperm whales in the action area are expected to be engaged in foraging during this time (Bando et al. 2010; Fujise et al. 2003; Tamura et al. 2005; Tamura et al. 2007; Tamura et al. 2006).

Table 22. Estimated exposure of ESA-listed whales to sound levels $\geq$160 dB re 1 µPa$_{rms}$ during the proposed seismic activities.

Whale density per 1,000 km^2	# of whales exposed to proposed activities	# of exposures to listed whales	Population size	% of population exposed	Population/location
NPRW-0.04	2	2	920	0.2	North Pacific[1]
Blue -0.40	4	4	4,900	0.08	North Pacific[2]
Fin -0.75	9	9	13,000	0.07	Sea of Okhotsk (study area)[1]
Sei -1.78	21	21	68,000	0.03	Western North Pacific[3]
Humpback -0.47	5	5	6,000-8,000	0.06	Western North Pacific stock[4]
Sperm -1.04	12	12	102,000	0.01	Western North Pacific[5]
Total	52	52	--	--	--

[1] Miyashita and Kato (1998)
[2] Gambell (1976)
[3] Hakamada et al. (2004)
[4] Calambokidis et al. (1997)
[5] Kato and Miyashita (2000)

Exposure of listed whales to multibeam echosounder and sub-bottom profiler. Two additional acoustic systems will operate during the proposed *Langseth* cruise: the multibeam echosounder and the sub-bottom profiler. Both of these systems have the potential to expose listed species to sound above the 160 dB re 1 µPa$_{rms}$ threshold. Both systems operate at generally higher frequencies than airgun operations (10.5-13 kHz for the multibeam echosounder and 3.5 kHz for the sub-bottom profiler) and this mitigates effects. As such, their frequencies will attenuate more rapidly than those from airgun sources. Listed individuals would experience higher levels of airgun noise well before either multibeam echosounder or sub-bottom profiler noise of equal amplitude would reach them. Thus, operational airguns mitigate multibeam echosounder and sub-bottom profiler noise exposure. While airguns are not operational, marine mammal observers will remain on duty to collect sighting data. If listed whales were to closely approach the vessel, the *Langseth* would take evasive actions to avoid a ship-strike and simultaneously mitigate exposure to very high source levels. As ship strike has already been ruled out as an insignificant effect, as well as high-level ensonification of listed whales (multibeam echosounder source level = 242 dB re 1 µPa$_{rms}$; sub-bottom profiler source level = 204 dB re 1 µPa$_{rms}$). Boebel et al. (2006) concluded that multibeam echosounders and sub-bottom profilers similar to those to be used during the proposed activities presented a low risk for auditory damage or any other injury, and that an individual would require exposure to 250–1,000 pulses from a sub-bottom profiler to be at risk for a temporary threshold shift (TTS). To be susceptible to TTS, a whale would have to pass at very close range and match the vessel's speed; we expect a very small probability of this during the proposed study. An individual would have to be well within 100 m of the vessel to experience a single multibeam echosounder pulse that could result in TTS (LGL Ltd. 2008). The same result could only occur at even closer ranges for sub-bottom profiler signals, because the signals are weaker. Furthermore, we expect both multibeam echosounder and sub-bottom profiler systems to operate continuously with duty cycles of 1-20 s. It is possible, however, that some small number of listed whales (fewer than those exposed to airguns) could experience low-level multibeam echosounder and/or sub-bottom profiler sound. We are unable to quantify the level of exposure.

Sea turtles

Exposure of listed turtles to airguns. The NSF did not quantify the number of exposure events, or the number of exposures per individual sea turtle. Green sea turtles, hawksbill sea turtles, leatherback sea turtles, loggerhead sea turtles, and olive ridley sea turtles may occur in the action area and, therefore, be exposed to airgun sound during the cruise. A combination of factors leads us to believe that each of these species will be present, although few direct sighting data exist for offshore marine waters of the northwestern Pacific. These factors include high-quality foraging habitat, nesting habitat for loggerhead and green sea turtles, and potential migratory corridors or developmental areas.

The action area borders a confluence of currents whose interaction results in highly-productive surface waters. As with listed whales, this area has the potential to draw large numbers of sea turtles to the area. Unfortunately, sea turtles are not as easy to detect from survey platforms as marine mammals and the lack of quantifiable data for the area is not surprising. Adult loggerhead sea turtles are known to reside in oceanic waters off Japan (Hatase et al. 2002b; Hatase and Sakamoto 2004; Hatase et al. 2002c). Juvenile loggerheads are known to forage in the Kuroshio Extension and productivity features associated with it (Polovina et al. 2004). Loggerhead occurrence in waters over the Shatsky Rise is bolstered by longline bycatch records (Lewison et al. 2004b; Yokota et al. 2006). Olive ridley and leatherback sea turtle species may also utilize the region for foraging (Komatsu et al. 2002; Polovina et al. 2006; Polovina et al. 2004; Polovina et al. 2003a; Polovina et al. 2000; Yokota et al. 2006). Neither species has been documented to be bycaught over the Shatsky Rise in longlines (Yokota et al. 2006).

In addition to foraging in the area, Japanese coasts serve as the primary site for loggerhead nesting in the North Pacific (Bowen et al. 1995; Kamezaki 1989; Kamezaki et al. 2003; Resendiz et al. 1998; Sea Turtle Association of Japan 2010; Uchida and Nishiwaki 1995). The peak of nesting season overlaps with the proposed time frame of the proposed seismic survey (Iwamoto et al. 1985). Females lay multiple nests per season and frequently travel offshore into the Kuroshio Current between nesting events, possibly to speed egg development (Eckert 1993b; Iwamoto et al. 1985; Nishimura 1994; NMFS and USFWS 1998d; Sato et al. 1998). Furthermore, hatchlings move into offshore waters, where the Kuroshio and Kuroshio Extension aide their movement to foraging habitat along Baja California (Bowen et al. 1995; Nichols 2005; Polovina et al. 2006; Polovina et al. 2000; Resendiz et al. 1998). Satellite telemetry and bycatch records indicate that individuals returning to Japanese shores to breed travel through waters over the Shatsky Rise, foraging along the way (Kobayashi et al. 2008; Nichols et al. 2000; Polovina et al. 2004).

Although not as extensive as loggerhead nesting, green sea turtles also nest on the Ogasawara Islands from May through September, peaking in June and July (Table 8); (Chan et al. 2007a; Eckert 1993b; Suganuma 1989; Suganuma et al. 1996; Tachikawa et al. 1994; Uchida and Nishiwaki 1995). Nesting also occurs along the Ryuuku Islands from May through August (Abe et al. 1998; Chan et al. 2007a; Kikukawa et al. 1996; Uchida 1994). Reproductive individuals likely return to nearshore waters along the Japanese mainland to forage, using nearshore waters where possible (Tachikawa et al. 1994; Uchida 1994; Uchida and Nishiwaki 1995). No green sea turtle bycatch has been documented over the Shatsky Rise (Yokota et al. 2006). Nesting has rarely or infrequently been documented for leatherback and hawksbill sea turtles, although leatherbacks are known to travel past Japan in association with reproductive migrations (Eckert 1993b; Kamezaki et al. 2002; Kikukawa et al. 1999; Sea Turtle Association of Japan 2010; Uchida and Nishiwaki 1995). Although documentation of green sea turtles in the action area is lacking, life history studies from other areas support juvenile green sea turtles leaving nesting beaches to develop in offshore areas such as those found over the Shatsky Rise (Limpus and Chaloupka 1997; Musick and Limpus 1997; NMFS and USFWS 1998a; Pelletier et al. 2003; Plotkin 2003).

Offshore Japanese waters may also serve as developmental habitat or migrations for leatherback, olive ridley, and hawksbill sea turtles (Kamezaki and Matsui 1997; NMFS and USFWS 1998c; Sea Turtle Association of Japan 2010; Seminoff et al. 2003b; Uchida 1994; Uchida and Nishiwaki 1995). This may be particularly true for olive ridleys, which use the North Pacific Gyre (which is partly bordered by the Kuroshio Current and Kuroshio Extension and in which the Shatsky Rise is located)(Polovina et al. 2006; Polovina et al. 2004; Polovina et al. 2003a). Satellite tracks during 2008 support the occurrence of adult female hawksbills in the action area during the seasonal time frame of the proposed survey (George Balazs, NMFS-PIFSC, personal communication).

Based upon this information, we expect exposure to all five sea turtle species. Exposure is likely least extensive for green sea turtles, whose habitats should restrict them to coastal waters, but may travel offshore. We also expect infrequent exposure to hawksbill sea turtles. A few satellite tracks as well as life history characteristics, such as juvenile use of offshore habitat as nurseries and adult use of current systems for interbreeding movements, makes it likely that hawksbills would be present in the area at the time of the survey (Kamezaki and Matsui 1997; NMFS and USFWS 1998c; NMFS and USFWS 2007d; Seminoff et al. 2003b; Uchida 1994; Uchida and Nishiwaki 1995). For these species, any age class or sex could be exposed, although more green sea turtle females may be exposed (sex ratios for this population area unknown, but other Pacific populations show a female bias). Leatherback and olive ridley sea turtles have not been documented to occur over the Shatsky Rise, but their life history patterns give us reason to believe that these species should be present. Again, either sex and any age class could be exposed, although juveniles may be the most extensively exposed age class for olive ridleys due to their use of the North Pacific Gyre for development. We expect exposure to more loggerhead sea turtles than any other species due to their extensive and multifaceted use of the region at various stages of their life history. We expect exposure to all sexes and life stages, including hatchlings and pregnant females.

Exposure of listed turtles to multibeam echosounder and sub-bottom profiler. As with baleen whales, sea turtles hear in the low frequency range. There is a low probability that sea turtles could experience exposure to sounds emitted by multibeam echosounder or sub-bottom profiler.

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 exposure 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.

Response of marine mammals 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 listed whales and sea turtles considered in this Opinion. Possible responses considered in this analysis consist of

- threshold shifts
- auditory interference (masking)
- behavioral responses
- 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 and sea turtles in the action area.

Marine mammals and threshold shifts. Exposure of marine mammals to very strong sound pulses can result in physical effects, such as changes to sensory hairs in the auditory system, which may temporarily or permanently impair hearing. Temporary threshold shifts (TTSs) can last minutes to days. Full recovery is expected and this condition is not considered a physical injury. However, a recent mouse study has shown that although full hearing can be regained from TTS (i.e., the sensory cells actually receiving sound are normal), damage can still occur to nerves of the cochlear nerve leading to delayed but permanent hearing damage (Kujawa and Liberman 2009). At higher received levels, or in frequency ranges where animals are more sensitive, permanent threshold shifts (PTSs) can occur in which auditory sensitivity is unrecoverable. Either of these conditions can result from a single pulse or from the accumulated effects of multiple pulses, in which case each pulse need not be as loud as a single pulse to have the same accumulated effect. TTS and PTS are specific only to the frequencies over which exposure occurs.

Few data are available to precisely define each listed species' hearing range, let alone its sensitivity and levels necessary to induce TTS or PTS. Based upon captive studies of odontocetes, our understanding of terrestrial mammal hearing, and extensive modeling, the best available information supports sound levels at a given frequency would need to be ~186 dB SEL or ~196-201 dB re 1 μPa_{rms} in order to produce a low-level TTS from a single pulse (Southall et al. 2007). If an individual experienced exposure to several airgun pulses of ~190 dB re 1 μPa_{rms}, PTS could occur. A marine mammal would have to be within 400 m of the *Langseth's* airgun array to be within the 190 dB re 1 μPa_{rms} isopleth and risk a TTS. PTS is expected at levels ~6 dB greater than TTS levels on a peak-pressure basis, or 15 dB greater on an SEL basis (Southall et al. 2007). Estimates that are conservative for species protection are 230 dB re 1 μPa (peak) for a single pulse, or multiple exposures to ~198 dB re 1 $\mu Pa^2 \cdot s$. In terms of exposure to the *Langseth's* airgun array, an individual would need to be within a few meters of the largest airgun to experience a single pulse >230 dB re 1 μPa peak (Caldwell and Dragoset 2000).

Overall, we do not expect TTS or PTS to occur to any listed whale as a result of airgun exposure for several reasons. We expect that individuals will move away from the airgun array as it approaches. We further believe that as sound intensity increases, individuals will experience conditions (stress, loss of prey, discomfort, etc.) that prompt them to move away from the sound source and thus avoid exposures that would induce TTS. Ramp-ups would also reduce the probability of TTS exposure at the start of seismic surveys. Furthermore, mitigation measures would be in place to initiate a ramp-down if individuals enter or are about to enter the 180 dB isopleth, which is below the levels believed to be necessary for potential TTS.

Marine mammals and auditory interference (masking). Interference, or masking, generally occurs when the interfering noise is of a similar frequency and similar to or louder than the auditory signal received by an animal processing echolocation signals or listening for acoustic information from other individuals. Masking can interfere with an individual's ability to gather acoustic information about its environment, such as predators, prey, conspecifics, and other environment cues. 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).

There is frequency overlap between airgun noise and vocalizations of listed whales, particularly baleen whales. 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). 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, masking should not be a concern in the proposed action. This is primarily because masking tends to result from continuous sounds rather than short pulses, such as seismic airguns (Richardson et al. 1995b). The *Langseth's* airguns will emit a 0.1 s pulse when fired every 15-73 s. Therefore, pulses will not "cover up" the vocalizations of listed whales to a significant extent (Madsen et al. 2002). We address the response of listed whales stopping vocalizations as a result of airgun sound in behavioral responses.

Marine mammals and behavioral responses. We expect the greatest response to airgun sounds by number and overall impact to be from behavioral responses. Listed individuals may briefly respond to underwater sound by slightly changing their behavior or relocating a short distance, in which case the effects are unlikely to be individually significant. Displacement from important feeding or breeding areas over a prolonged period would likely be significant. This has been suggested for humpback whales along the Brazilian coast as a result of increased seismic activity (Parente et al. 2007). Marine mammal responses to anthropogenic sound vary by species, state of maturity, prior exposure, current activity, reproductive state, time of day, and other factors (Ellison et al. 2012). Although some studies are available which address responses of listed whales considered in this opinion directly, additional studies to other related whales (such as bowhead and gray whales) are relevant in determining the responses expected by species under consideration. Therefore, studies from non-listed or species outside the action area are also considered here. Individual differences in responding to stressful stimuli also appear to exist and appear to have at least a partial genetic basis in trout (Laursen et al. 2011). Animals generally respond to anthropogenic perturbations as they would predators, increasing vigilance and altering habitat selection (Reep et al. 2011).

Several studies have aided in assessing the various levels at which whales may modify or stop their calls in response to airgun sound. Whales continue calling while seismic surveys are occurring locally (Greene Jr et al. 1999; Jochens et al. 2006; Madsen et al. 2002; McDonald et al. 1993; McDonald et al. 1995; Nieukirk et al. 2004; Richardson et al. 1986; Smultea et al. 2004; Tyack et al. 2003). Some blue, fin, and sperm whales stopped calling for short and long periods apparently in response to airguns (Bowles et al. 1994; Clark and Gagnon 2006; McDonald et al. 1995). Fin whales (presumably adult males) engaged in singing in the Mediterranean Sea moved out of the area of a seismic survey while airguns were operational as well as for at least a week thereafter (Castellote et al. 2012). A blue whale discontinued calls in response to received airgun sound of 143 dB re 1 μPa for 1 hour before resuming (McDonald et al. 1995). Blue whales may instead attempt to compensate for elevated ambient sound by calling more frequently during seismic surveys (Iorio and Clark 2009). Sperm whales, at least under some conditions, may be particularly sensitive to airgun sounds, as they have been documented to cease calling in association with airguns being fired hundreds of kilometers away (Bowles et al. 1994). Other studies have found no response by sperm whales to received airgun sound levels up to 146 dB re 1 μPa_{p-p} (Madsen et al. 2002; McCall Howard 1999). Some exposed individuals may cease calling in response to the *Langseth's* airguns. If individuals ceased calling in response to the *Langseth's* airguns during the course of the proposed survey, the effect would likely be temporary.

There are numerous studies of the responses of some baleen whale to airguns, although responses to lower-amplitude sounds are known, most studies seem to support a threshold of ~160 dB re 1 μPa_{rms} as the received sound level to cause behavioral responses other than vocalization changes (Richardson et al. 1995c). Activity of individuals seems to influence response, as feeding individuals seem to respond less than mother/calf pairs and migrating individuals (Harris et al. 2007; Malme and Miles 1985; Malme et al. 1984; Miller et al. 1999; Miller et al. 2005; Richardson et al. 1995c; Richardson et al. 1999b). Migrating bowhead whales show strong avoidance reactions to received 120–130 dB re 1 μPa_{rms} exposures at distances of 20-30 km, but only changed dive and respiratory patterns while feeding and showed avoidance at higher received sound levels (152–178 dB re 1 μPa_{rms}; (Harris et al. 2007; Ljungblad et al. 1988; Miller et al. 1999; Miller et al. 2005; Richardson et al. 1995c; Richardson et al. 1999b; Richardson et al. 1986). Responses such as stress may occur and the threshold for displacement may simply be higher while feeding. Bowhead calling rate was found to decrease during migration in the Beaufort Sea as well as temporary displacement from seismic sources (Nations et al. 2009). Despite the above information and exposure to repeated seismic surveys, bowheads continue to return to summer feeding areas and when displaced, bowheads appear to reoccupy areas within a day (Richardson et al. 1986). We do not know whether the individuals exposed in these ensonified areas are the same returning or whether individuals that tolerate repeat exposures may still experience a stress response.

Gray whales respond similarly. Gray whales discontinued feeding and/or moved away at received sound levels of 163 dB re 1 μPa_{rms} (Bain and Williams 2006; Gailey et al. 2007; Johnson et al. 2007a; Malme and Miles 1985; Malme et al. 1984; Malme et al. 1986; Malme et al. 1988; Würsig et al. 1999; Yazvenko et al. 2007a; Yazvenko et al. 2007b). Migrating gray whales began to show changes in swimming patterns at ~160 dB re 1 μPa and slight behavioral changes at 140-160 dB re 1 μPa_{rms} (Malme and Miles 1985; Malme et al. 1984). As with bowheads, habitat continues to be used despite frequent seismic survey activity, but long-term effects have not been identified, if they are present at all (Malme et al. 1984). Johnson et al. (2007b) 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.

Humpback whales continue a pattern of lower threshold of response when not occupied with feeding. Migrating humpbacks altered their travel path (at least locally) along Western Australia at received levels as low as 140 dB re 1 μPa_{rms} when females with calves were present, or 8-12 km from the seismic source (McCauley et al. 2000a; McCauley et al. 1998). A startle response occurred as low as 112 dB re 1 μPa_{rms}. Closest approaches were generally limited to 3-4 km, although some individuals (mainly males) approached to within 100 m on occasion where sound levels were 179 dB re 1 μPa_{rms}. Changes in course and speed generally occurred at estimated received level of 157–164 dB re 1 μPa_{rms}. Feeding humpbacks appear to be somewhat more tolerant. Humpback whales along Alaska startled at 150–169 dB re 1 μPa and no clear evidence of avoidance was apparent at received levels up to 172 re 1 μPa_{rms} (Malme et al. 1984; Malme et al. 1985). Potter et al. (2007) found that humpbacks on feeding grounds in the Atlantic did exhibit localized avoidance to airguns. Among humpback whales on Angolan breeding grounds, no clear difference was observed in encounter rate or point of closest approach during seismic versus non-seismic periods (Weir 2008).

Observational data are sparse for specific baleen whale life history data (breeding and feeding grounds). Available data support a general avoidance response. Some fin and sei whale sighting data indicate similar sighting rates during seismic versus non-seismic periods, but sightings tended to be further away and individuals remained underwater longer (Stone 2003; Stone and Tasker 2006). Other studies have found at least small differences in sighting rates (lower during seismic activities) as well as whales being more distant during seismic operations (Moulton et al. 2006a; Moulton et al. 2006b; Moulton and Miller 2005). When spotted at the average sighting distance, individuals would have likely been exposed to ~169 dB re 1 μPa_{rms} (Moulton and Miller 2005).

Sperm whale response to airguns has thus far included mild behavioral disturbance (disrupted foraging, avoidance, cessation of vocal behavior) or no reaction. Several studies have found Atlantic sperm whales to show little or no response (Davis et al. 2000b; Madsen et al. 2006; Miller et al. 2009; Moulton et al. 2006a; Moulton and Miller 2005; Stone 2003; Stone and Tasker 2006; Weir 2008). Detailed study of Gulf of Mexico sperm whales suggests some alteration in foraging from <130-162 dB re 1 μPa_{p-p}, although other behavioral reactions were not noted by several authors (Gordon et al. 2004; Jochens et al. 2006; Madsen et al. 2006; Winsor and Mate 2006). This has been contradicted by other studies, which found avoidance reactions by sperm whales in the Gulf of Mexico in response to seismic ensonification (Jochens and Biggs 2004; Mate et al. 1994). Johnson and Miller (2002) noted possible avoidance at received sound levels of 137 dB re 1 μPa. Other anthropogenic sounds, such as pingers and sonars, disrupt behavior and vocal patterns (Goold 1999; Watkins et al. 1985; Watkins and Schevill 1975). Miller et al. (2009) found sperm whales to be generally unresponsive to airgun exposure in the Gulf of Mexico, with possible but inconsistent responses that included delayed foraging and altered vocal behavior. Displacement from the area was not observed. The lack of response by this species may in part be due to its higher range of hearing sensitivity and the low-frequency (generally <188 Hz) pulses produced by seismic airguns (Richardson et al. 1995c). Sperm whales are exposed to considerable energy above 500 Hz (Goold and Fish 1998). Breitzke et al. (2008) found that source levels were ~30 dB re 1 μPa lower at 1 kHz and 60 dB re 1 μPa lower at 80 kHz compared to dominant frequencies during a seismic source calibration. 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).

For whales exposed to seismic airguns during the proposed activities, behavioral changes stemming from airgun exposure may result in loss of feeding opportunities. We expect listed whales exposed to seismic airgun sound will exhibit an avoidance reaction, displacing individuals from the area. We also expect secondary foraging areas to be available that whales could continue feeding. In addition, we expect exposure of a given area to be brief and reoccupation can occur soon after the *Langseth* transects through. Although breeding may be occurring, we are unaware of any habitat features that sperm whales would be displaced from if sperm whales depart an area as a consequence of the *Langseth*'s presence. We expect breeding may be temporarily disrupted if avoidance or displacement occurs, but we do not expect the loss of any breeding opportunities are expected.

Marine mammals and physical or physiological effects. Individual whales exposed to airguns (as well as other sound sources) could experience effects not readily observable, such as stress, that can significantly affect life history.

Stress is an adaptive response and does not normally place an animal at risk. Distress involves a stress response resulting in a biological consequence to the individual. The mammalian stress response involves the hypothalamic-pituitary-adrenal (HPA) axis being stimulated by a stressor, causing a cascade of physiological responses, such as the release of the stress hormones cortisol, adrenaline (epinephrine), glucocorticosteroids, and others (Busch and Hayward 2009)(Gulland et al. 1999; Morton et al. 1995; St. Aubin and Geraci 1988; St. Aubin et al. 1996; Thomson and Geraci 1986). These hormones subsequently can cause short-term weight loss, the liberation of glucose into the blood stream, impairment of the immune and nervous systems, elevated heart rate, body temperature, blood pressure, and alertness, and other responses (Busch and Hayward 2009; NMFS 2006g)(Cattet et al. 2003; Delehanty and Boonstra 2009; Elftman et al. 2007; Fonfara et al. 2007; Kaufman and Kaufman 1994; Mancia et al. 2008; Moe and Bakken 1997; Noda et al. 2007; Thomson and Geraci 1986)(Dierauf and Gulland 2001; Omsjoe et al. 2009). In some species, stress can also increase an individual's susceptibility to gastrointestinal parasitism (Greer et al. 2008). In highly-stressful circumstances, or in species prone to strong "fight-or-flight" responses, more extreme consequences can result, including muscle damage and death (Cowan and Curry 1998; Cowan and Curry 2002; Cowan and Curry 2008; Herraez et al. 2007). The most widely-recognized indicator of vertebrate stress, cortisol, normally takes hours to days to return to baseline levels following a significantly stressful event, but other hormones of the HPA axis may persist for weeks (Dierauf and Gulland 2001). Mammalian stress levels can vary by age, sex, season, and health status (Gardiner and Hall 1997; Hunt et al. 2006; Keay et al. 2006; Kenagy and Place 2000; Nunes et al. 2006; Romero et al. 2008; St. Aubin et al. 1996). Stress is lower in immature right whales than adults and mammals with poor diets or undergoing dietary change tend to have higher fecal cortisol levels (Hunt et al. 2006; Keay et al. 2006; Kitaysky and Springer 2004).

Loud noises generally increase stress indicators in mammals and fishes (Kight and Swaddle 2011). Romano et al. (2004) found beluga whales and bottlenose dolphins exposed to a seismic water gun (up to 228 dB re 1 $\mu\text{Pa} \cdot \text{m}_{p-p}$) and single pure tones (up to 201 dB re 1 μPa) had increases in stress chemicals, including catecholamines, which could affect an individual’s ability to fight off disease. During the time following September 11, 2001, shipping traffic and associated ocean noise decreased along the northeastern U.S.; this decrease in ocean noise was associated with a significant decline in fecal stress hormones in North Atlantic right whales, providing evidence that chronic exposure to increased noise levels, although not acutely injurious, can produce stress (Rolland et al. 2012). These levels returned to baseline after 24 hours. As whales use hearing as a primary way to gather information about their environment and for communication, we assume that limiting these abilities would be stressful. Stress responses may also occur at levels lower than those required for TTS (NMFS 2006g). Therefore, exposure to levels sufficient to trigger onset of PTS or TTS are expected to be accompanied by physiological stress responses (NMFS 2006g; NRC 2003). As we do not expect individuals to experience TTS or PTS, (see *Marine mammals and threshold shifts*), we also do not expect any listed individual to experience a stress response at high levels. We assume that a stress response could be associated with displacement or, if individuals remain in a stressful environment, the stressor (sounds associated with the airgun, multibeam echosounder, or sub-bottom profiler) will dissipate in a short period as the vessel (and stressors) transects away without significant or long-term harm to the individual via the stress response.

Exposure to loud noise can also adversely affect reproductive and metabolic physiology (Kight and Swaddle 2011). Premature birth and indicators of developmental instability (possibly due to disruptions in calcium regulation) have been found in embryonic and neonatal rats exposed to loud sound. In fish eggs and embryos exposed to sound levels only 15 dB greater than background, increased mortality was found and surviving fry had slower growth rates (a similar effect was observed in shrimp), although the opposite trends have also been found in sea bream. Dogs exposed to loud music took longer to digest food. The small intestine of rats leaks additional cellular fluid during loud sound exposure, potentially exposing individuals to a higher risk of infection (reflected by increases in regional immune response in experimental animals). Exposure to 12 hours of loud noise can alter elements of cardiac tissue. In a variety of factors, including behavioral and physiological responses, females appear to be more sensitive or respond more strongly than males (Kight and Swaddle 2011). It is noteworthy that although various exposures to loud noise appear to have adverse results, exposure to music largely appears to result in beneficial effects in diverse taxa; the impacts of even loud sound are complex and not universally negative (Kight and Swaddle 2011).

Marine mammals and strandings. There is some concern regarding the coincidence of marine mammal strandings and proximal seismic surveys. No conclusive evidence exists to causally link stranding events to seismic surveys.

Suggestions that there was a link between seismic surveys and strandings of humpback whales in Brazil (Engel et al., 2004) were not well founded (IAGC, 2004; IWC, 2007). In September 2002, two Cuvier's beaked whales stranded in the Gulf of California, Mexico. The *R/V Ewing* had been operating a 20-airgun, 8,490-in^3 airgun array 22 km offshore the general area at the time that strandings occurred. The link between the stranding and the seismic surveys was inconclusive and not based on any physical evidence (Hogarth, 2002; Yoder, 2002) as some vacationing marine mammal researchers who happened upon the stranding were ill-equipped to perform an adequate necropsy. Furthermore, the small numbers of animals involved and the lack of knowledge regarding the spatial and temporal correlation between the beaked whales and the sound source underlies the uncertainty regarding the linkage between seismic sound sources and beaked whale strandings (Cox et al., 2006).

Responses of marine mammal prey. Seismic surveys may also have indirect, adverse effects on prey availability through lethal or sub-lethal damage, stress responses, or alterations in their behavior or distribution. Studies described herein provide extensive support for this, which is the basis for later discussion on implications for listed whales. Unfortunately, species-specific information on the prey of listed whales is not generally available. Until more specific information is available, we expect that teleost, cephalopod, and krill prey of listed whales to react in manners similar to those described herein.

Some support has been found for fish or invertebrate mortality resulting from airgun exposure, and this is limited to close-range exposure to high-amplitudes (Bjarti 2002; Falk and Lawrence 1973; Hassel et al. 2003; Holliday et al. 1987; Kostyuchenko 1973; La Bella et al. 1996; McCauley et al. 2000a; McCauley et al. 2000b; McCauley et al. 2003; Popper et al. 2005; Santulli et al. 1999). Lethal effects, if any, are expected within a few meters of the airgun array (Buchanan et al. 2004). We expect fish to be capable of moving away from the airgun array if it causes them discomfort.

More evidence exists for sub-lethal effects. Several species at various life stages have been exposed to high-intensity sound sources (220-242 dB re 1 μPa) at close distances, with some cases of injury (Booman et al. 1996; McCauley et al. 2003). TTS was not found in whitefish at received levels of ~175 dB re 1 $\mu Pa^2 \cdot s$, but pike did show 10-15 dB of hearing loss with recovery within 1 day (Popper et al. 2005). Caged pink snapper have experienced PTS when exposed over 600 times to received seismic sound levels of 165-209 dB re 1 μPa_{p-p}.

By far the most common response by fishes is a startle or distributional response, where fish react momentarily by changing orientation or swimming speed, or change their vertical distribution in the water column. Startle responses were observed in rockfish at received airgun levels of 200 dB re 1 μPa_{0-p} and alarm responses at >177 dB re 1 μPa_{0-p} (Pearson et al. 1992). Fish also tightened schools and shifted their distribution downward. Normal position and behavior resumed 20-60 minutes after seismic firing ceased. A downward shift was also noted by Skalski et al. (1992) at received seismic sounds of 186–191 re 1 μPa_{0-p}. Caged European sea bass showed elevated stress levels when exposed to airguns, but levels returned to normal after 3 days (Skalski et al. 1992). These fish also showed a startle response when the survey vessel was as much as 2.5 km away; this response increased in severity as the vessel approached and sound levels increased, but returned to normal after ~2 hours following cessation of airgun activity. Whiting exhibited a downward distributional shift upon exposure to 178 dB re 1 μPa_{0-p} airgun sound, but habituated to the sound after 1 hour and returned to normal depth (sound environments of 185-192 dB re 1 μPa) despite airgun activity (Chapman and Hawkins 1969).

Whiting may also flee from airgun sound (Dalen and Knutsen 1986). Hake may redistribute downward (La Bella et al. 1996). Lesser sandeels exhibited initial startle responses and upward vertical movements before fleeing from the survey area upon approach of an active seismic vessel (Hassel et al. 2003; Hassel et al. 2004). McCauley et al. (2000; 2000a) found smaller fish show startle responses at lower levels than larger fish in a variety of fish species and generally observed responses at received sound levels of 156–161 dB re 1 μPa_{rms}, but responses tended to decrease over time suggesting habituation. As with previous studies, caged fish showed increases in swimming speeds and downward vertical shifts. Pollock did not respond to airgun sounds received at 195–218 dB re 1 μPa_{0-p}, but did exhibit continual startle responses and fled from the seismic source when visible (Wardle et al. 2001). Blue whiting and mesopelagic fishes were found to redistribute 20–50 m deeper in response to airgun ensonification and a shift away from the survey area was also found (Slotte et al. 2004). Salmonid swim bladders were reportedly damaged by received sound levels of ~230 dB re 1 μPa (Falk and Lawrence 1973). Startle responses were infrequently observed from salmonids receiving 142–186 dB re 1 μPa_{p-p} sound levels from an airgun (Thomsen 2002). Cod and haddock likely vacate seismic survey areas in response to airgun activity and estimated catchability decreased starting at received sound levels of 160–180 dB re 1 μPa_{0-p} (Dalen and Knutsen 1986; Engås et al. 1996; Engås et al. 1993; Løkkeborg 1991; Løkkeborg and Soldal 1993; Turnpenny et al. 1994). Bass did not appear to vacate during a shallow-water seismic survey with received sound levels of 163–191 dB re 1 μPa_{0-p} (Turnpenny and Nedwell 1994). Similarly, European sea bass apparently did not leave their inshore habitat during a 4-5 month seismic survey (Pickett et al. 1994).

Squid responses to airguns have also been studied, although to a lesser extent than fishes. In response to airgun exposure, squid exhibited both startle and avoidance responses at received sound levels of 174 dB re 1 μPa_{rms} by first ejecting ink and then moving rapidly away from the area (McCauley et al. 2000a; McCauley et al. 2000b). The authors also noted some movement upward. During ramp-up, squid did not discharge ink but alarm responses occurred when received sound levels reached 156–161 dB re 1 μPa_{rms}

The overall response of fishes and squids is to exhibit startle responses and undergo vertical and horizontal movements away from the sound field. We do not expect krill (the primary prey of most listed baleen whales) to experience effects from airgun sound. Although humpback whales consume fish regularly, we expect that any disruption to their prey will be temporary, if at all. Therefore, we do not expect any adverse effects from lack of prey availability to baleen whales. Sperm whales regularly feed on squid and some fishes and we expect individuals to feed while in the action area during the proposed survey. Based upon the best available information, fishes and squids ensonified by the ~160 dB isopleths could vacate the area and/or dive to greater depths, and be more alert for predators. We do not expect indirect effects from airgun activities through reduced feeding opportunities sufficient to reach a significant level. Effects are likely to be temporary and, if displaced, both sperm whales and their prey would re-distribute back into the area once survey activities have passed.

Marine mammal response to multibeam echosounder and sub-bottom profiler. We expect listed whales to experience ensonification from not only airguns, but also seafloor mapping systems. Multibeam echosounder and sub-bottom profiler frequencies are much higher than frequencies used by all listed whales except humpback and sperm whales. Although Todd et al. (1992) found that mysticetes reacted to sonar sounds at 3.5 kHz within the 80-90 dB re 1 μPa range, it is difficult to determine the significance of this because the source was a signal designed to be alarming and the sound level was well below typical ambient noise. Hearing is poorly understood for listed baleen whales, but it is assumed that they are most sensitive to frequencies over which they vocalize, which are much lower than frequencies emitted by the multibeam echosounder and sub-bottom profiler systems (Ketten 1997; Richardson et al. 1995c). Thus, if blue, fin, sei, or North Pacific right whales are exposed, they are unlikely to hear these frequencies well (if at all) and a response is not expected.

Assumptions for humpback and sperm whale hearing are much different than for other listed whales. Humpback and sperm whales vocalize between 3.5-12.6 kHz and an audiogram of a juvenile sperm whale provides direct support for hearing over this entire range (Au 2000; Au et al. 2006; Carder and Ridgway 1990; Erbe 2002a; Frazer and Mercado 2000; Goold and Jones 1995; Levenson 1974; Payne and Payne 1985; Payne 1970; Richardson et al. 1995c; Silber 1986; Thompson et al. 1986; Tyack 1983; Tyack and Whitehead 1983; Weilgart and Whitehead 1993; Weilgart and Whitehead 1997; Weir et al. 2007; Winn et al. 1970). Maybaum (1990; 1993) observed that Hawaiian humpbacks moved away and/or increased swimming speed upon exposure to 3.1-3.6 kHz sonar. 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. Sperm whales have stopped vocalizing in response to 6-13 kHz pingers, but did not respond to 12 kHz echo-sounders (Backus and Schevill 1966; Watkins 1977; Watkins and Schevill 1975).

We do not expect masking of sperm or humpback whale communications to appreciably occur due to multibeam echosounder or sub-bottom profiler signal directionality, low duty cycle, and the brief period when an individual could be within its beam.

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. The sonars proposed for use by L-DEO differ from sonars used during naval operations, which generally have a longer pulse duration and more horizontal orientation than the more downward-directed multibeam echosounder and sub-bottom profiler. The sound energy received by any individuals exposed to the multibeam echosounder and sub-bottom profiler sources during the proposed activities is lower relative to naval sonars, as is the duration of exposure. The area of possible influence for the multibeam echosounder and sub-bottom profiler is also much smaller, consisting of a narrow zone close to and below the source vessel. Although navigational sonars are operated routinely by thousands of vessels around the world, stranding incidence has been correlated to use of these sonars. Because of these differences, we do not expect these systems to contribute to a stranding event.

Sea turtles

Sea turtle response to airguns. As with marine mammals, sea turtles may experience

- threshold shifts
- behavioral responses
- non-auditory physical or physiological effects

Sea turtles and threshold shifts. Few data are available to assess sea turtle hearing, let alone the effects seismic equipment may have on their hearing potential. The only study which addressed sea turtle TTS was conducted by Moein et al. (1994), in which a loggerhead experienced TTS upon multiple airgun exposures in a shallow water enclosure, but recovered within 1 day.

Although data on the precise levels that can result in TTS or PTS are lacking, we do not expect either of these to occur to any sea turtle as a result of the proposed action. As with marine mammals, we assume that sea turtles will not move towards a source of stress or discomfort. Some experimental data suggest sea turtles may avoid seismic sources (McCauley et al. 2000a; McCauley et al. 2000b; Moein et al. 1994), but monitoring reports from seismic surveys in other regions suggest that some sea turtles do not avoid airguns and were likely exposed to higher levels of seismic airgun pulses (Smultea and Holst 2003). For this reason, mitigation measures are also in place to limit sea turtle exposure. We do not expect reduction in foraging opportunities by the proposed action.

Sea turtles and behavioral responses. As with listed whales, it is likely that sea turtles will experience behavioral responses in the form of avoidance. O'Hara and Wilcox (1990) found loggerhead sea turtles exhibited an avoidance reaction at an estimated sound level of 175–176 dB re 1 µPa $_{\text{rms}}$ (or slightly less) in a shallow canal (McCauley et al. 2000a). Green and loggerhead sea turtles avoided airgun sounds at received sound levels of 166 dB re 1 µPa and 175 dB re 1 µPa, respectively (McCauley et al. 2000a; McCauley et al. 2000b). Sea turtle swimming speed increased and becomes more erratic at 175 dB re 1 µPa, with individuals becoming agitated. Loggerheads also appeared to move towards the surface upon airgun exposure (Lenhardt 1994; Lenhardt et al. 1983). Recent monitoring studies show that some sea turtles move away from approaching airguns, although sea turtles may approach active seismic arrays within 10 m (Holst et al. 2006; LGL Ltd 2005a; LGL Ltd 2005b; LGL Ltd 2008; NMFS 2006e; NMFS 2006h).

Observational evidence suggests that sea turtles are not as sensitive to sound as are marine mammals and behavioral changes are expected when sound levels rise above received sound levels of 166 dB re 1 µPa. This corresponds with previous reports of sea turtle hearing thresholds being generally higher than for marine mammals (DFO 2004). At 166 dB re 1 µPa, we anticipate some change in swimming patterns and a stress response of exposed individuals. Some turtles may approach the active seismic array to closer proximity, but we expect them to eventually turn away. We expect temporary displacement of exposed individuals from some portions of the action area while the *Langseth* transects through. We are aware of a single stranding event associated with a seismic survey involving 30 dead sea turtles (Jaszy and Horowitz 2005). Evidence linking the survey with the stranding is inconclusive and characteristics of that survey (shallow nearshore waters) are dissimilar to the proposed survey. We do not expect lethal effects on sea turtles for the proposed survey or an appreciable reduction in their feeding potential.

Sea turtles and stress. Direct evidence of seismic sound causing stress is lacking in sea turtles. However, sea turtles actively avoid high-intensity exposure to airguns in a fashion similar to predator avoidance. As predators generally induce a stress response in their prey (Dwyer 2004; Lopez and Martin 2001; Mateo 2007), we assume that sea turtles experience a stress response to airguns when they exhibit behavioral avoidance or when they are exposed to sound levels apparently sufficient to initiate an avoidance response (~166 dB re 1 µPa). We expect breeding adult females may experience a lower stress response, as female loggerhead, hawksbill, and green sea turtles appear to have a physiological mechanism to reduce or eliminate hormonal response to stress (predator attack, high temperature, and capture) in order to maintain reproductive capacity at least during their breeding season; a mechanism apparently not shared with males (Jessop 2001; Jessop et al. 2000; Jessop et al. 2004). Individuals may experience a stress response at levels lower than ~166 dB re 1 µPa, but data are lacking to evaluate this possibility.

Response of sea turtles to multibeam echosounder and subbottom profiler. Sea turtles do not possess a hearing range that includes frequencies emitted by these systems. Therefore, listed sea turtles will not hear these sounds even if they are exposed and are not expected to respond to them.

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.

We expect that those aspects described in the *Environmental baseline* will continue to impact listed resources into the foreseeable future. We expect climate change, ship-strikes, bycatch, and harvests to continue into the future. Movement towards bycatch reduction and greater foreign protections of sea turtles are generally occurring through the Pacific Ocean, which may aid in abating the downward trajectory of sea turtle populations.

Integration and synthesis of effects

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 (Anderson 2000; Brandon 1978; Mills and Beatty 1979; Stearns 1992). As a result, if the assessment indicates that 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) to which those individuals belong, and then to the species those population(s) represent.

Listed whales. The NSF proposes to fund a seismic survey by L-DEO that could incidentally harass several listed whale species. These species include: blue whales, fin whales, humpback whales, sei whales, North Pacific right whales, and sperm whales, all of whom are endangered throughout their ranges.

The *Status of listed resources* section identified commercial whaling as the primary reason for reduced populations, many of whom are a small fraction of their former abundance (Tables 3-7). Although large-scale commercial harvests no longer occur for these species, some harvests from subsistence and scientific research in regional and worldwide populations still occur. Other worldwide threats to the survival and recovery of listed whale species include: altered prey base and habitat quality as a result of global warming, ship strike, entanglement in fishing gear, toxic chemical burden and biotoxins, ship noise, competition with commercial fisheries, and killer whale predation. Populations of whales inhabiting the northwestern Pacific face area-specific threats identified in the *Environmental baseline*, including whaling.

Despite these pressures, available trend information indicates most local populations of listed whales are stable or increasing (Tables 3-7). As previously mentioned, the *Cumulative effects* section identifies actions in the *Environmental baseline* we expect to generally continue for the foreseeable future.

The *Effects analysis* supports the conclusion of harassment to listed whales by proposed seismic activities. We expect individuals and exposure to be four blue, 9 fin, 21 sei, five humpback, two North Pacific right, and 12 sperm whales to airgun sounds which will elicit a behavioral response of temporarily moving out of the area. We expect a low-level, transitory stress response to accompany this behavior. The number of individuals exposed represent a tiny fraction of the populations (not in excess of 0.2%; Tables 3-7) and reactions should not limit the fitness of any single individual. The other actions we considered in the Opinion, the operation of multibeam echosounder and sub-bottom profiler systems, are not expected to be audible to blue, fin, sei or North Pacific right whales and consequently are not expected to have any direct effects on this species. However, humpback and sperm whales could hear sounds produced by these systems. Responses could include cessation of vocalization by sperm whales and/or movement out of the survey area by both species. We do not expect these effects to have fitness consequences for any individual. The *Effects analysis* also found that, although sperm whales may experience temporarily reduced feeding opportunities, this indirect effect would be transient and not reduce individual fitness of any whale. Overall, we do not expect a fitness reduction to any individual whale. As such, we do not expect fitness consequences to populations or listed whale species as a whole.

Listed turtles. Listed turtles that are expected to occur within the action area include green sea turtles, hawksbill sea turtles, leatherback sea turtle, loggerhead sea turtles, and olive ridley sea turtles, which are either threatened or endangered. The *Status of listed resources* section found that most sea turtle populations have undergone significant to severe reduction by human harvesting of both eggs and turtles, as well as severe bycatch pressure in worldwide fishing industries. As previously mentioned, the *Cumulative effects* section identified actions in the *Environmental baseline* (including bycatch, harvest, and climate change) to generally continue for the foreseeable future.

From the *Effects analysis*, we expect that green, hawksbill, leatherback, loggerhead, and olive ridley sea turtles could experience exposure to airgun sounds and be harassed by these sounds. These sounds may induce a temporary effect in low-level stress levels, swimming patterns, and movement out of the action area. Population size is not available to calculate the subset of each population affected. Data were not available to calculate the number of exposures, but we do not expect the number of ensonifications to alter critical life functions. We do expect transient responses that do not affect the fitness of any one individual. We do not expect impairment of local nesting by the proposed survey. As we do not expect any sea turtle to be capable of hearing signals produced by the multibeam echosounder and sub-bottom profiler systems, we do not expect direct effects from these systems on sea turtle fitness. We do not anticipate any indirect effects from the proposed actions to influence sea turtles. Overall, we do not expect any individual sea turtle to undergo a fitness consequence.

Conclusion

After reviewing the current status of blue, fin, sei, humpback, North Pacific right, and sperm whales, as well as green, hawksbill, leatherback, loggerhead, and olive ridley sea turtles; the

Environmental baseline for the action area; the anticipated effects of the proposed activities; and the *Cumulative effects*, it is the NMFS' Opinion that the actions (NSF's funding of and the Permits and Conservation Division's issuance of an IHA for seismic surveys over the Shatsky Rise) are not likely to jeopardize the continued existence of these species. Similarly, it is the NMFS' Opinion that the issuance of an IHA by the NMFS' Permits and Conservation Division for harassment that would occur incidental to the proposed action is not likely to jeopardize the continued existence of these species. No critical habitat co-occurs within the action area and thus the proposed action would have no effect on critical habitat.

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 the NMFS as an act which actually kills or injures wildlife, which may include significant habitat modification or degradation which actually kills or injures fish or wildlife 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 NSF and the Permits and Conservation Division so that they become binding conditions for L-DEO 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, the NMFS will issue a statement that specifies the impact of any incidental taking of endangered or threatened species. To minimize such impacts, reasonable and prudent measures and terms and conditions to implement the measures, must be provided. Only incidental take resulting from the agency actions and any specified reasonable and prudent measures and terms and conditions identified in the incidental take statement are exempt from the taking prohibition of Section 9(a), pursuant to Section 7(o) of the ESA.

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 Permits and Conservation Division's proposed authorization of the incidental taking of fin, blue, sei, humpback, North Pacific right, and sperm whales pursuant to Section 101(a)(5)(D) of the MMPA. 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.

Amount or extent of take

The NMFS anticipates the proposed seismic survey in the Pacific Ocean over the Shatsky Rise might result in the incidental take of listed species. The proposed action is expected to take four blue, nine fin, 21 sei, five humpback, two North Pacific right, and 12 sperm whales by exposing individuals to received seismic sound levels greater than 160 dB re 1 μPa by harassment. These estimates are based on the best available information of whale densities in the area to be ensonified above 160 dB re 1 μPa during the proposed activities. This incidental take would result primarily from exposure to acoustic energy during seismic operations and 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 expect the proposed action will also take individual 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 sea turtles is expected to occur at received levels above 166 dB re 1 μPa. As we cannot determine the number of individuals to which harassment will occur, we expect the extent of exposure will occur within the 166 dB isopleth of the *Langseth's* airgun array.

Harassment of blue, fin, humpback, North Pacific right, sei, and sperm whales exposed to seismic studies at levels less than 160 dB re 1 μPa, or of green, hawksbill, leatherback, loggerhead, and olive ridley sea turtles at levels less than 166 dB re 1 μPa, is not expected. If overt adverse reactions (for example, startle responses, dive reactions, or rapid departures from the area) by listed whales or sea turtles are observed outside of the 160 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 airguns, multibeam echosounder, or sub-bottom profiler are in operation, this may constitute take that is not covered in this Incidental Take Statement. The NSF and the Permits and Conservation Division must contact the Endangered Species Act Interagency Cooperation Division to determine whether reinitiation of consultation is required because of such operations.

Any incidental take of blue, fin, humpback whales, North Pacific right, sei whales, sperm whales, or green sea turtles, hawksbill sea turtles, leatherback sea turtles, loggerhead sea turtles, and olive ridley sea turtles is restricted to the permitted action as proposed. If the actual incidental take meets or exceeds the predicted level, the NSF and Permits and Conservation Division must reinitiate consultation. All anticipated takes would be "takes by harassment", as described previously, involving temporary changes in behavior.

Reasonable and prudent measures

The NMFS believes the reasonable and prudent measures described below are necessary and appropriate to minimize the impact of incidental take of listed whales and sea turtles resulting from the proposed action. These measures are non-discretionary and must be binding conditions of the NSF funding of the proposed seismic studies and the NMFS' authorization for the exemption in Section 7(o)(2) to apply. If the NSF or the NMFS fail to ensure compliance with these terms and conditions, the protective coverage of Section 7(o)(2) may lapse.

1. For listed sea turtle and marine mammal species these measures include the following: immediate shutdown of all seismic sources in the event a North Pacific right whale is detected; vessel-based visual monitoring by marine mammal and sea turtle observers; real-time passive acoustic monitoring by marine mammal and sea turtle observers; speed or course alteration as practicable; implementation of a marine mammal and sea turtle exclusion zone within the 180 dB re 1 μPa_{rms} isopleth for power-down and shut-down procedures; emergency shutdown procedures in the event of an injury or mortality of a listed marine mammal or sea turtle; and ramp-up procedures when starting up the array. The measures for marine mammals are required to be implemented through the terms of the IHA issued under section

101(a)(5)(D) and 50 CFR 216.107.

2. The implementation and effectiveness of mitigation measures incorporated as part of the Reasonable and Prudent Measure mentioned above and the associated Terms and Conditions must be monitored.

Terms and conditions

In order to be exempt from the prohibitions of Section 9 of the ESA, the NSF, Permits and Conservation Division, and L-DEO must comply with the following terms and conditions, which implement the Reasonable and Prudent Measures described above. These terms and conditions are non-discretionary.

To implement the Reasonable and Prudent Measures, the NSF and the NMFS shall ensure that

1. L-DEO implements the mitigation, monitoring, and reporting conditions contained in the IHA and this Opinion.
2. The Chief of the Endangered Species Act Interagency Cooperation Division is immediately informed of any changes or deletions to any portions of the monitoring plan or IHA.
3. L-DEO immediately reports all sightings and locations of injured or dead endangered and threatened species to the Permits and Conservation Division and NSF.
4. The NSF and the Permits and Conservation Division provide a summary of the implementation and effectiveness of the terms of the IHA to the Chief of the Endangered Species Act Interagency Cooperation Division. This report shall confirm the implementation of each term and summarize the effectiveness of the terms for minimizing the adverse effects of the project on listed whales and sea turtles.

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. *Effects of seismic noise on sea turtles.* The NSF should promote and fund research examining the potential effects of seismic surveys on listed sea turtle species.

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

Reinitiation notice

This concludes formal consultation on the proposed seismic source survey to be funded by the NSF and conducted by the L-DEO on board the *R/V Langseth* in the Pacific Ocean over the Shatsky Rise, and the issuance of an incidental harassment authorization for the proposed studies pursuant to Section 101(a)(5)(D) of the Marine Mammal Protection Act (MMPA). As provided in 50 CFR §402.16, consultation must be reinitiated if control over the action has been retained (or is authorized by law) and if: (1) the amount or extent of incidental take is exceeded; (2) new information reveals effects of the agency action that may affect listed species or critical habitat in a manner or to an extent not considered in this opinion; (3) the agency action is subsequently modified in a manner that causes an effect to the listed species or critical habitat not considered in this opinion; or (4) a new species is listed or critical habitat designated that may be affected by the action. In instances where the amount or extent of authorized take is exceeded, Section 7 consultation must be reinitiated immediately.

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