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

Biological Opinion

Agencies: United States Geological Survey and NOAA's National Marine Fisheries Service-Office of Protected Resources-Permits and Conservation Division

Activities Considered: Low energy seismic survey by the United States Geological Survey in the Gulf of Mexico 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: Perry Gayaudo For Helen Golde

Date: 4.16.13

Section 7(a)(2) of the Endangered Species Act (ESA)(16 U.S.C. 1531 *et seq.*) requires that each federal agency 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 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 activities described in this document, the Federal action agencies are the United States Geological Survey (USGS), which proposes to conduct using an academic seismic survey vessel, the *Pelican*, to seismically survey in offshore waters of the Gulf of Mexico from 15 April to 9 June 2013 and the NMFS' Office of Protected Resources-Permits and Conservation Division, which proposes to authorize the USGS to "take" marine mammals incidental to this seismic survey. The consulting agency is the NMFS' Office of Protected Resources – Endangered Species Act Interagency Cooperation Division.

This document represents the NMFS- Endangered Species Act Interagency Cooperation Division's 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(IHA) application, draft IHA, environmental assessment, monitoring reports from similar activities, published and unpublished scientific information on endangered and threatened species and their surrogates, scientific and commercial information such as reports from government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.

BIOLOGICAL OPINION

Consultation History

-On 1 August 2012, the USGS requested a list of resources potentially overlapping with the proposed action. On the same date, the Endangered Species Act Interagency Cooperation Division provided a comprehensive table of listed resources under NMFS' jurisdiction that were expected to be affected by the proposed action.

-On 5 November 2012, the NMFS' Permits and Conservation Division (Permits Division) received an application for the USGS to incidentally harass marine mammal and sea turtle species during a seismic survey cruise in the Gulf of Mexico.

-On 10 December 2012, the NMFS' Endangered Species Act Interagency Cooperation Division received a request for formal consultation from the USGS to incidentally harass marine mammal and sea turtle species during the same cruise.

-On 13 February 2013, the Endangered Species Act Interagency Cooperation Division received a request for formal consultation from the Permits Division to authorize incidental harassment of marine mammals during the proposed cruise. Information was suifficient to initiate consultation on this date.

-On 20 February 2013, the NMFS' Permits Division published a notice in the Federal Register of its proposal to issue an IHA for the proposed action and solicitation of public comment on the action.

Description of the Proposed Action

USGS Seismic Survey

The USGS proposes to conduct a marine seismic survey in the Gulf of Mexico from 15 April to 4 May 2013. The IHA is proposed to be effective until 9 June 2013.

Marine seismic surveys will be conducted by the *R/V Pelican* (*Pelican*). The planned seismic survey will consist of approximately 1,480 km of survey lines (some of which may be re-shot), all in water 1,500-2,000 m deep. An array of two airguns, totaling 210 in^3 in discharge volume, will be deployed as an energy source. A 450 m-long hydrophone streamer will also be deployed. Twenty-five ocean bottom seismometers (OBSs) will be deployed. The Permits 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 develop technology and to collect data to assist in the characterization of marine gas hydrates in order to better understand their potential as an energy resource, their impact on seafloor stability, and their role in climate change.

The survey would occur exclusively in the territorial seas of the U.S. south of Louisiana, although not in the waters under any State jurisdiction. All planned geophysical data acquisition activities will be conducted by USGS and in coordination with on-board assistance of the scientists (Drs. Haines and Hart of the USGS) who have proposed the study.

Schedule

The *Pelican* is due to sail from Cocodrie, Louisiana on or about 16 April to the action area (Figure 1). The USGS proposes to conduct roughly four days of airgun activities at each of two sites. After this time, the *Pelican* will return to port in Cocodrie, Louisiana on or about 4 May.

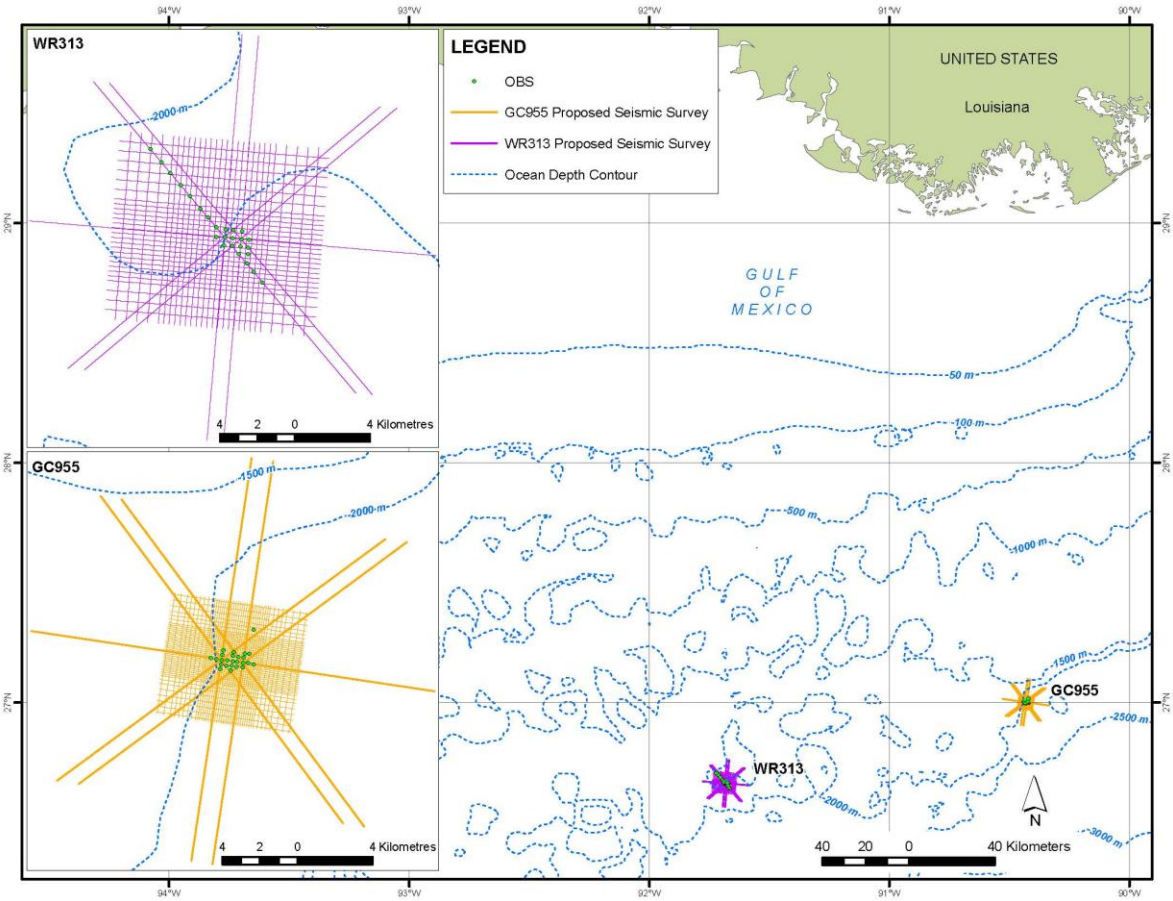

Figure 1. Proposed area for the marine seismic survey in the Gulf of Mexico from 15 April to 9 June 2013.

Source Vessel Specifications

The *Pelican* (or, possibly, another similar vessel) will tow the two-airgun array along predetermined lines (Figure 1). The operating speed during seismic acquisition is typically about 8.1 km/h. When not towing seismic survey gear, the *Pelican* typically cruises at 17 km/h. Due to the relatively short length of equipment being towed behind the vessel, the turning radius of the *Pelican* will not be limited and turning to avoid collisions or approaches to protected species will occur. The *Pelican* will also serve as the platform from which marine mammal and sea turtle observers (PSVOs) would watch for animals.

Airgun Description

The airgun array will consist of two Sercel generator-injector airguns, with a total volume of approximately 210 in^3. Although initial information indicated that a sparker might also be used, this device will not be employed. The two airguns will be towed approximately 21 m behind the

vessel. The tow depth of the array will be 3 m. The airgun array will fire roughly every 6-10 s. During firing, a brief pulse of sound will be emitted, but be silent during the intervening periods. This signal attenuates as it moves away from the source, decreasing in amplitude, but also increasing in signal duration. Airguns will operate continually while at each of the two sites, but be shut down during intervening times as well as during transit to and from sites.

Two-airgun array specifications

- Energy source — Two GI airguns of 105 in^3 each
- Source output (downward) — 0-pk is 5.5 bar-m (234.4 dB re 1 μPa·m); pk-pk is 9.8 bar·m (239.8 dB)
- Air discharge volume — ~210 in^3
- Dominant frequency components — 0–188 Hz

Because the actual source originates from two 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 will be substantially lower than the nominal source level applicable to downward propagation because of the directional nature of the sound from the airgun array.

Ocean Bottom Seismometers

At each of the two study sites, 25 OBSs will be deployed and retrieved by the *Pelican*. Once ready for retrieval, an acoustic release transponder would interrogate the ocean bottom seismometer at a frequency of 9–11 kHz, and the *Pelican* would receive a response at a frequency of 9–13 kHz. The burn wire release assembly would then activate, and the instrument would release from the anchor and float to the surface.

Predicted Sound Levels vs. Distance and Depth

The Lamont-Doherty Earth Observatory (L-DEO) has predicted received sound levels, in relation to distance and direction from two 105-in^3 GI airguns in deep water (Fig. 3). Empirical data concerning 190, 180, and 160 dB re 1 μPa_{rms} distances were acquired during the acoustic calibration study of the *Langseth*'s airgun array in a variety of configurations in 2007-2008 (Tolstoy et al. 2009). As a two-airgun array was not measured, the estimates provided here were extrapolated from other results using conservative assumptions. 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) from in the proposed survey (3 m). Because propagation varies with array depth, correction factors have been applied to the distances reported by Tolstoy et al. (2009).

Table 1 shows the distances at which four rms (root mean squared) sound levels are expected to be received from the airgun array. The 190 dB re 1 μPa_{rms} distance is the safety criteria as specified by NMFS (1995) for cetaceans. The 180 dB distance will 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 two-airgun array.

Source and volume	**Tow depth (m)**	**Predicted rms radii (m)**			
		190 dB	**180 dB**	**170 dB**	**160 dB**
Two GI airguns 210 in^3	3	20	70	~205	670

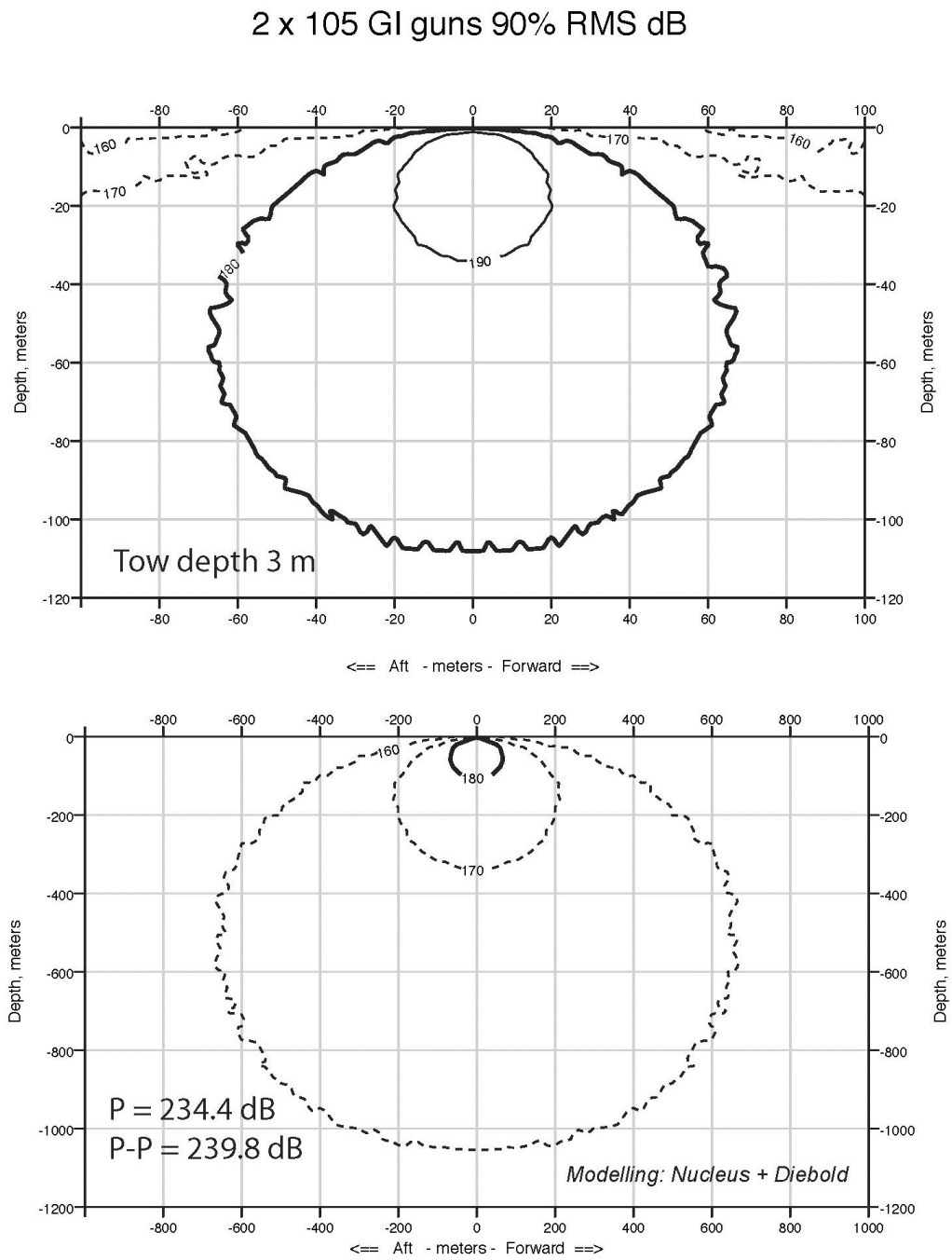

Figure 3. Modeled received sound levels (SELs) from two 105-in^3 GI airgun operating in deep water at a three meter tow depth. Received rms levels (SPLs) are likely ~10 dB higher.

NMFS Permits Division's Proposed Incidental Harassment Authorization

The NMFS' Permits 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 will be valid from 15 April through 9 June 2013, and will authorize the incidental harassment of the following endangered species (among other species): sperm whales (*Physeter macrocephalus*) and other non-listed marine mammals. The proposed IHA identifies the following requirements that USGS must comply with as part of its authorization.

A. Establish a safety radius corresponding to the anticipated 180-dB isopleth for full (210 in^3) airgun operations.

B. Use one, NMFS-approved, vessel-based protected species visual observer (PSVO) to watch for and monitor marine mammals near the seismic source vessel and accessory vessels during daytime airgun operations, start-ups of airguns at night, and while the seismic array and streamers are 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. Laser range finding binoculars (Leica LRF 1200 laser rangefinder or equivalent) will be available to assist with distance estimation. PSVOs shifts will last no longer than 4 hours at a time. PSVOs 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 PSVOs, for at least 30 min prior to starting the airgun (day or night). If PSVOs find a marine mammal within the safety zone, USGS must delay the seismic survey until the marine mammal has left the area. If the PSVO sees a marine mammal that surfaces, then dives below the surface, the observer shall wait 30 minutes. If the PSVO 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 (e.g. 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, USGS 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. Apply a “ramp-up” procedure when starting up at the beginning of seismic operations or any time after the entire array has been shut down for more than 15 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. During ramp-up, the PSVOs 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.

F. 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.

G. Shut-down or power-down the airguns upon marine mammal detection within, approaching, or entering the safety radius. 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).

H. To the maximum extent practicable, schedule seismic operations (i.e., shooting airguns) during daylight hours and OBS operations (i.e., deploy/retrieve) during nighttime hours. Marine seismic surveys may continue into night and low-light hours if such segment(s) of the survey is initiated when the entire relevant exclusion zones are visible and can be effectively monitored. No initiation of airgun array operations is permitted from a shut-down position at night or during low-light hours (such as in dense fog or heavy rain) when the entire relevant exclusion zone cannot be effectively monitored by the PSVO(s) on duty.

I. 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, USGS will immediately cease operating all authorized sound sources and report the incident to the Chief of the Permits Division, Office of Protected Resources, NMFS, at 301-427-8401. USGS will postpone the research activities until NMFS is able to review the circumstances of the take. NMFS will work with USGS to determine whether modifications in the activities are appropriate and necessary, and notify USGS that they may resume the seismic survey operations.

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

K. Conduct seismic operations during daylight hours where possible.

L. USGS 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 USGS 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) for cetaceans and 190 dB re 1 microPa (rms) for pinnipeds with a discussion of any specific behaviors those individuals exhibited.

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) and 190 dB re 1 microPa (rms) for pinnipeds with a discussion of the nature of the probable consequences of that exposure on the individuals that have been exposed.

iv. A description of the implementation and effectiveness of the:

a. Terms and conditions of the Opinion's Incidental Take Statement.

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 E*xposure 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 Division when it initiates formal consultation, the general scientific literature, and expert opinion.

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 Bureau of Ocean Energy Management, 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 are 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 determined 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 and predictive environmental factors, 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 (e.g.: blue, fin, 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 will in the northwestern Gulf of Mexico between coordinates 25.5-27.0° N and 90.5-92.0° W and between 15 April and 9 June 2013 (Figure 1). The survey would encompass water from 1,500-2,000 m deep in an area of 2,479 km^2. Responses to seismic sound sources by listed species occur within the 160 dB isopleths (modeled to be 640 m from the *Pelican*). This expands the action area beyond the seismic survey track lines (~1,480 km) to an ensonified region of 2,479 km^2.

Status of Listed Resources

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*.).

Table 2. Listed species in the action area. Asterisks denote species with critical habitat in the action area

Common Name (Distinct Population Segment, Evolutionarily Significant Unit, or Subspecies)	***Scientific Name***	***Status***
Cetaceans		
Sperm whale	*Physeter macrocephalus*	Endangered
Marine Turtles		
Leatherback sea turtle	*Dermochelys coriacea*	Endangered
Loggerhead sea turtle – Northeast Atlantic DPS	*Caretta caretta*	Threatened

Kemp's sea turtles may occupy offshore waters of the Gulf of Mexico. However, we do not expect occurrence during the time of the proposed action. Adults migrate to nesting beaches along a coastal corridor and breed in nearshore waters. These areas will not co-occur with the proposed action. Hatchlings nesting from beaches will not be present until after the proposed seismic survey has been completed. Immature individuals tend to reside in coastal foraging habitats, which also will not be exposed to components of the proposed action. Therefore, we discount the possibility of Kemp's ridley sea turtle exposure during the proposed action.

Although green sea turtles do occur in the Gulf of Mexico, they generally remain coastal in nature where they can find abundant foraging and breeding habitat (Landry and Costa 1999; Meylan et al. 1995; NMFS and USFWS 1991; USAF 1996). However, excursions into offshore waters are known. Hawksbill sea turtles are known to occur in the Gulf of Mexico, but generally remain along Florida. However, strandings along the Texas coast are known (Amos 1989; Collard and Ogren 1990; Hildebrand 1983; Landry and Costa 1999). Although these species generally do not ingest pelagic prey, they have not been documented as pelagic longline bycatch in the Gulf of Mexico (Garrison 2003a; Garrison 2005; Walsh and Garrison 2006). In addition, data from observers in the Gulf of Mexico indicate Kemp's ridley, green, and hawksbill occur over shelf waters and not in deeper waters (Greg Gitschlagg, NMFS, pers. comm., 2013). Based upon these data, we discount the potential for green and hawksbill sea turtle exposure to the

proposed seismic survey.

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.

Cetaceans

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). However, vocal dialects indicate parent-offspring transmission that indicates differentiation in populations (Rendell et al. 2011). 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). Sperm whales of the Gulf of Mexico may exists as a discrete stock based upon movement, genetic, and photoidentification studies (Jochens et al. 2008). Genetic studies indicate that movements of both sexes through expanses of ocean basins are common, and that males, but not females, often breed in different ocean basins than the ones in which they were born (Whitehead 2003). Sperm whale populations appear to be structured socially, at the level of the clan, rather than geographically (Whitehead 2003; Whitehead et al. 2008).

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 off 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 are found throughout the Mid-Atlantic Bight. Distribution extends further northward to areas north of Georges Bank and the Northeast Channel region in summer and then south of New England in fall, back to the Mid-Atlantic Bight. In the eastern Atlantic, 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 40° N in winter (Gosho et al. 1984a; Miyashita et al. 1995 as cited in Carretta et al. 2005; Rice 1974). Sperm whales are found year-round in Californian and Hawaiian waters (Barlow 1995; Dohl et al. 1983b; Forney et al. 1995; Lee 1993; Mobley Jr . et al. 2000; Rice 1960; Shallenberger 1981), but they reach peak abundance from April through mid-June and from the end of August through mid-November (Rice 1974). They are seen in every season except winter (December-February) off Washington and Oregon (Green et al. 1992). Summer/fall surveys in the eastern tropical Pacific (Wade and Gerrodette 1993) show that although sperm whales are widely distributed in the tropics, their relative abundance tapers off markedly towards the middle of the tropical Pacific and northward towards the tip of Baja California (Carretta et al. 2006). Sperm whales occupying the California Current region are genetically distinct from those in the eastern tropical Pacific and Hawaiian waters (Mesnick et al. 2011). The discreteness of the latter two areas remains uncertain (Mesnick et al. 2011).

Mediterranean. Sperm whales are found 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 50º and decrease in proportion to males south of 46-47º.

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). However, no sperm whale in the Pacific has been known to travel to points over 5,000 km apart and only rarely have been known to move over 4,000 km within a time frame of several years. This means that although sperm whales do not appear to cross from eastern to western sides of the Pacific (or vice-versa), significant mixing occurs that can maintain genetic exchange. Movements of several hundred kilometers are common, (i.e. between the Galapagos Islands and the Pacific coastal Americas). Movements appear to be group or clan specific, with some groups traveling straighter courses than others over the course of several days. However, general transit speed averages about 4 km/h. Sperm whales in the Caribbean region appear to be much more restricted in their movements, with individuals repeatedly sighted within less than 160 km of previous sightings.

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. In the Gulf oif Mexico, sperm whales appear to prefer deeper waters

(Baumgartner et al. 2001; Davis et al. 2002b) with high site fidelity (Jochens et al. 2008; Weller et al. 2000). While deep water is their typical habitat, sperm whales are rarely found in waters less than 300 m in depth (Clarke 1956; Rice 1989a). Off California, most sightings were in deep water near the continental shelf edge and none were observed during a summer survey through the action area in 2011 (Dohl et al. 1983a; NSF 2012). 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. 2000b; Davis et al. 2000c; Davis et al. 2000d; Davis et al. 2002a; Wormuth et al. 2000). Surface waters with sharp horizontal thermal gradients, such as along the Gulf Stream in the Atlantic, may also be temporary feeding areas for sperm whales (Griffin 1999; Jaquet et al. 1996; Waring et al. 1993). Sperm whales appear to be particularly concentrated and recurrent over Mississippi Canyon (Biggs et al. 2005; Mullin et al. 1991a; Mullin and Hoggard 2000; Mullin et al. 1991b; Würsig et al. 2000b).

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. 2000a). 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 1978). Estimated annual mortality rates of sperm whales are thought to vary by age, but previous estimates of mortality rate for juveniles and adults are now considered unreliable (IWC 1980). In addition to anthropogenic threats, there is evidence that sperm whale age classes are subject to predation by killer whales (Arnbom et al. 1987; Pitman et al. 2001).

Stable, long-term associations among females form the core of sperm whale societies (Christal et al. 1998). Up to about a dozen females usually live in such groups, accompanied by their female and young male offspring. Young individuals are subject to alloparental care by members of either sex and may be suckled by non-maternal individuals (Gero et al. 2009). Group sizes may be smaller overall in the Caribbean Sea (6-12 individuals) versus the Pacific (25-30 individuals)(Jaquet and Gendron 2009a). In the Gulf of California (the closest location to the

action area where sperm whale group size has been assessed robustly), groups contained an average of 24.7 individuals (Jaquet and Gendron 2009b). Groups may be stable for long periods, such as for 80 days in the Gulf of California (Jaquet and Gendron 2009a). 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 down 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. 2000a). Sperm whales typically travel ~3 km horizontally and 0.5 km vertically during a foraging dive (Whitehead 2003). Differences in night and day diving patterns are not known for this species, but, like most diving air-breathers for which there are data (rorquals, fur seals, and chinstrap penguins), sperm whales probably make relatively shallow dives at night when prey are closer to the surface.

Feeding. Sperm whales appear to feed regularly throughout the year (NMFS 2006). 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. Sound production and reception by sperm whales are better understood than in most cetaceans. Sperm whales produce broad-band clicks in the frequency range of 100 Hz to 20 kHz that can be extremely loud for a biological source (200-236 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; Norris and Harvey. 1972). Long, repeated clicks are associated with feeding and echolocation (Goold and Jones 1995; Weilgart and Whitehead 1993; Weilgart and Whitehead 1997). However, clicks are also used in short patterns (codas) during social behavior and 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. However, behavioral responses of adult, free-ranging individuals also provide insight into hearing range; sperm whales have been observed to frequently stop echolocating in the presence of underwater pulses made by echosounders and submarine sonar (Watkins et al. 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 3 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 2003).

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 data to determine population trends (Waring et al. 2008). Sperm whale were widely harvested from the northeastern Caribbean (Romero et al. 2001) and the Gulf of Mexico where sperm whale fisheries operated during the late 1700s to the early 1900s (NMFS 2006; Townsend 1935). The best estimate of the Gulf oif Mexico population is 763 individuals (NMFS 2012).

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. 2007). 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 2003).

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. Sperm whales are also hunted for subsistence purposes by whalers from Lamalera, Indonesia, where a traditional whaling industry has been reported to kill up to 56 sperm whales per year.

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 2003). Whitehead (2002b) estimated 12,069 sperm whales south of 60° S.

Table 3. 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. 2000a)
	--	1,110,000	672,000-1,512,000	360,000	105,984-614,016*	(Whitehead 2002)
North Atlantic	Basinwide	224,800	--	22,000	--	(Gosho et al. 1984a; Würsig et al. 2000a)
	Northeast Atlantic, Faroes-Iceland, and U.S. East Coast (combined)	--	--	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	(NMFS 2008)
	Northern Gulf of Mexico - off the Mississippi River Delta between 86 ^{o} and 91 ^{o}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	--	472,100	--	(Gosho et al. 1984a)
				930,000	--	(Rice 1989a)
	Eastern Tropical Pacific	--	--	26,053	13,797-38,309*	(Whitehead 2003)
	Off Costa Rica	--	--	1,360	823-2,248*	(Gerrodette and Palacios 1996)
	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 - North Pacific stock	--	--	--	--	(Angliss and Allen 2007)
	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)
Southern Hemisphere	Basinwide	547,600	--	299,400	--	(Gosho et al. 1984a; 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	17,613-238,387*	(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).

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 of strandings 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). However, other estimates have included 436,000 individuals killed between 1800-1987 (Carretta et al. 2005). However, all of these estimates are likely underestimates due to illegal and inaccurate killings by Soviet whaling fleets between 1947-1973. In the Southern Hemisphere, these whalers killed an estimated 100,000 whales that they did not report to the IWC (Yablokov et al. 1998), with smaller harvests in the Northern Hemisphere, primarily the North Pacific, that extirpated sperm whales from large areas (Yablokov and Zemsky 2000). Additionally, Soviet whalers disproportionately killed adult females in any reproductive condition (pregnant or lactating) as well as immature sperm whales of either gender.

Following a moratorium on whaling by the IWC, significant whaling pressures on sperm whales were eliminated. However, sperm whales are known to have become entangled in commercial fishing gear and 17 individuals are known to have been struck by vessels (Jensen and Silber 2004). Japan maintains an active whaling fleet, killing up to 10 sperm whales annually (IWC 2008). In 2009, one sperm whale was killed during western North Pacific surveys (Bando et al. 2010).

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 2003). A sperm whale was struck and died just east of the action area in September 2003 (Brownell Jr. et al. 2009).

Whale-watching vessels are known to influence sperm whale behavior (Richter et al. 2006). An individual was caught and released from gillnetting, although injured, on Georges Bank during 1990. A second individual was freed, but injured, from gillnetting on George's Bank in 1995. In 1994, a sperm whale was disentangled from gillnet along the coast of Maine.
In August 1993, a dead sperm whale, with longline gear wound tightly around the jaw, was found floating ~32 km off Maine.

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

Critical Habitat. The NMFS has not designated critical habitat for sperm whales.

Sea Turtles

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. 2006). 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/Guianas, 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 2006b; 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. 2006).

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

Nesting beaches also occur in Mexico and Costa Rica (nesting occurs October through March) are a separate population from the western Pacific beaches (Benson et al. 2007a; summary in NMFS and USFWS 2007d; Spotila 2004b). 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.

Leatherbacks are the most frequently-bycaight sea turtle species in the pelagic Gulf of Mexico (Garrison 2003b; Garrison 2005; Walsh and Garrison 2006). Although none have been observed in the action area that we know of, several summer sightings have occurred (USN 2007).

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).

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.

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 1984).

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 if their prey (Harvey et al. 2006). Leatherback prey in the Gulf of Alaska are frequently concentrated in the deep-scattering layer (Hodge and Wing 2000). Mean dive and surface durations were 2.9 and 2.2 min, respectively (Harvey et al. 2006). In a study comparing diving patterns during foraging versus travelling, leatherbacks dove shallower (mean of 53.6 m) and moved more slowly (17.2 km/day) while in foraging areas while travelling to or from these areas (81.8 m and 51.0 km/day)(Fossette et al. 2009b).

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; NMFS 2001b).

Nesting aggregations occur along Gabon, Sao Tome and Principe, French Guiana, Suriname, and Florida (Bräutigam and Eckert 2006b; 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 2001b). 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/Guianas 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 (Putrawidjaja 2000; Suárez et al. 2000) (Dutton et al. 2007). 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 is equally severe. Nesting at Terengganu, Malaysia is 1% of that in 1950s (Chan and Liew 1996). The South China Sea and East Pacific nesting colonies have undergone catastrophic collapse. Overall, Pacific populations have declined from an estimated 81,000 individuals to <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. 2009b).

Anthropogenic Threats. Leatherback nesting and marine environments are facing increasing impacts through widespread development and tourism along nesting beaches (Hamann et al. 2006; Hernandez et al. 2007; Maison 2006; Santidrián Tomillo et al. 2007). Structural impacts to beaches include building and piling construction, beach armoring and renourishment, and sand extraction (Bouchard et al. 1998; Lutcavage et al. 1997b). 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). Along the coast of Peru, 13% of 140 leatherback carcasses were found to contain plastic bags and film (Fritts 1982). 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. 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. An estimated 6,363 leatherback sea turtles were documented as caught by the U.S. Atlantic tuna and swordfish longline fisheries between 1992-1999 (NMFS 2001a). 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 2004). In 2010, there were 26 observed interactions between leatherback sea turtles and longline gear used in the HMS fishery (Garrison and Stokes 2011). All leatherbacks were released alive, with all gear removed for the majority of captures. While 2010 total estimates are not yet available, in 2009, 285.8 (95% CI: 209.6-389.7) leatherback sea turtles are estimated to have been taken in the longline fisheries managed under the HMS FMP based on the observed takes (Garrison and Stokes 2011). Lewison et al. (2004b) 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 f40 mortalities) and 100 interactions in U.S. Pacific fisheries (resulting in about 10 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 a total of 37 leatherbacks were incidentally captured (16 lethally) in drift gillnets set in offshore waters from Maine to Florida during this period. Observer coverage for this period ranged from 54 to 92%. Trinidad and Tobago's Institute for Marine Affairs estimated that more than 3,000 leatherbacks were captured incidental to gillnet fishing in the coastal waters of Trinidad in 2000. Half or more of the gravid turtles in Trinidad and Tobago waters may be killed (Lee Lum 2003), though many of the turtles do not die as a result of drowning, but rather because the fishermen butcher them in order to get them out of their nets (NMFS 2001a).

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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 and Oregon (Cape Flattery to Cape Blanco; 64,760 km^2) and California (Point Arena to Point Arguello; 43,798 km^2). The primary constituent element of these areas includes the occurrence of prey species, primarily scyphomedusae of the order Semaeostomeae (*Chrysaora, Aurelia, Phacellophora,* and *Cyanea*), of sufficient condition, distribution, diversity, abundance and density necessary to support individual as well as population growth, reproduction, and development of leatherbacks. It is worth noting that the foraging period (spring through early summer) does not occur when the action is proposed to occur.

Loggerhead Sea Turtle-Northwestern Atlantic DPS

Distribution. Loggerheads are circumglobal occurring throughout the temperate and tropical regions of the Atlantic, Pacific, and Indian oceans. Loggerheads are the most abundant species of sea turtle found in U.S. coastal waters.

Population Designations. Five groupings represent loggerhead sea turtles by major sea or ocean basin: Atlantic, Pacific, and Indian oceans, as well as Caribbean and Mediterranean seas. As with other sea turtles, populations are frequently divided by nesting aggregation (Hutchinson and Dutton 2007). On September 22, 2011, the NMFS designatde 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 (76 FR 58868).

Atlantic Ocean. Western Atlantic nesting locations include The Bahamas, Brazil, and numerous locations from the Yucatán Peninsula to North Carolina (Addison 1997; Addison and Morford 1996; Marcovaldi and Chaloupka 2007). This group comprises five nesting subpopulations: Northern, Southern, Dry Tortugas, Florida Panhandle, and Yucatán. Additional nesting occurs on Cay Sal Bank (Bahamas), Cuba, the Bahamian Archipelago, Quintana Roo (Yucatan Peninsula), Colombia, Brazil, Caribbean Central America, Venezuela, and the eastern Caribbean Islands. Genetic studies indicate that, although females routinely return to natal beaches, males may breed with females from multiple populations and facilitate gene flow Bowen et al. (2005). In the eastern Atlantic, we know of five rookeries from Cape Verde, Greece, Libya, Turkey, and the western Africa coast. The northwestern Atlantic DPS is considered to be bounded by the equator and 60º N latitude and extend east to 40º W in the Atlantic basin; this is based upon oceanographic features satellite telemetry, sightings, and bycatch data (Conant et al. 2009).

Reproduction and Growth. Loggerhead nesting is confined to lower latitudes temperate and subtropic zones but absent from tropical areas (NMFS and USFWS 1991b; NRC 1990b; 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 2005). 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 1985; NMFS 2001a; Witherington et al. 2006).

Loggerhead mating likely occurs along migration routes to nesting beaches, as well as in 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).

Loggerhead sea turtles belonging to the northwestern Atlantic DPS nest along the shoreline from Virginia to Alabama primarily from April to September (peaking in June and July), but can also include the Bahamas, the Yucatan Peninsula of Mexico, and southwestern Cuba and other locations along Central and South America (Conant et al. 2009). Once hatched, individuals disperse to convergence zones, driftlines, and areas covered by *Sargassum* weed (particularly the North Atlantic Gyre), presumably for the foraging and shelter these areas occur to hatchlings (Conant et al. 2009).

Migration and Movement. 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 2001a). For individuals of the northwestern Atlantic DPS, this includes areas along the U.S. Atlantic coast, the Bahamas, Mexico, Cuba, and the Antilles (Conant et al. 2009). Adult loggerheads make lengthy migrations from nesting beaches to foraging grounds (TEWG 1998a). In the Gulf of Mexico, larger females tend to disperse more broadly after nesting than smaller individuals, which tend to stay closer the nesting location (Girard et al. 2009). In the North Atlantic, loggerheads travel north during spring and summer as water temperatures warm and return south in fall and winter, but occur offshore year-round assuming adequate temperature. For immature individuals, this movement occurs in two patterns: a north-south movement over the continental shelf with migration south of Cape Hatteras in winter and movement north along Virginia for summer foraging, and a not-so-seasonal oceanic dispersal into the Gulf Stream as far north as the 10-15˚ C isotherm (Mansfield et al. 2009). Wallace et al. (2009) suggested differences in growth rate based upon these foraging strategies. There is conflicting evidence that immature loggerheads roam the oceans in currents and eddies and mix from different natal origins or distribute on a latitudinal basis that corresponds with their natal beaches (Monzon-Arguello et al. 2009; Wallace et al. 2009). McCarthy et al. (2010) found that movement patterns of loggerhead sea turtles were more convoluted when sea surface temperatures were higher, ocean depths shallower, ocean currents stronger, and chlorophyll a levels lower.

Sighting and stranding records support loggerhead sea turtles to be common, year-round residents of the Gulf of Mexico, although their abundance is much greater in the northeastern region versus the northwestern (Davis et al. 2000a; Fritts et al. 1983; Landry and Costa 1999). Loggerheads may occur in both offshore habitats (particularly around oil platforms and reefs, where prey and shelter are available; (Davis et al. 2000a; Fritts et al. 1983; Gitschlag and Herczeg 1994; Lohoefener et al. 1990; Rosman et al. 1987), as well as shallow bays and sounds (which may be important developmental habitat for late juveniles in the eastern Gulf of Mexico; (Davis et al. 2000a; Lohoefener et al. 1990; USAF 1996). Offshore abundance in continental slope waters increases during the winter in the eastern Gulf of Mexico, as cooler inshore waters force individuals into warmer offshore areas (Davis et al. 2000a). Although none have been observed in the action area that we know of, several summer sightings have occurred nearby (USN 2007).

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

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. 2002). 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. 2003; 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). The NMFS recently reclassified loggerhead turtles in the western North Atlantic Ocean as endangered (76 FR 58868).

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).

Atlantic Ocean. The greatest concentration of loggerheads occurs in the Atlantic Ocean and the adjacent Caribbean Sea, primarily on the Atlantic coast of Florida, with other major nesting areas located on the Yucatán Peninsula of Mexico, Columbia, Cuba, South Africa (EuroTurtle 2006 as cited in LGL Ltd. 2007; Márquez 1990).

Among the five subpopulations, loggerhead females lay 53,000-92,000 nests per year in the southeastern U.S. and the Gulf of Mexico, and the total number of nesting females is 32,000-56,000. All of these are currently in decline or data are insufficient to access trends (NMFS 2001a; TEWG 1998b). Loggerheads from western North Atlantic nesting aggregations may or may not feed in the same regions from which they hatch. Loggerhead sea turtles from the northern nesting aggregation, which represents about 9% of the loggerhead nests in the western North Atlantic, comprise 25-59% of individuals foraging from Georgia up to the northeast U.S. (Bass et al. 1998; Norrgard 1995; Rankin-Baransky 1997; Sears 1994; Sears et al. 1995). Loggerheads associated with the South Florida nesting aggregation occur in higher frequencies

in the Gulf of Mexico (where they represent ~10% of the loggerhead captures) and the Mediterranean Sea (where they represent ~45% of loggerhead sea turtles captured). About 4,000 nests per year are laid along the Brazilian coast (Ehrhart et al. 2003).

Because of its size, the south Florida subpopulation of loggerheads may be critical to the survival of the species in the Atlantic, and in the past it was considered second in size only to the Oman nesting aggregation (NMFS 2006e; NMFS and USFWS 1991b). The South Florida population increased at ~5.3% per year from 1978-1990, and was initially increasing at 3.9-4.2% after 1990. An analysis of nesting data from 1989-2005, a period of more consistent and accurate surveys than in previous years, showed a detectable trend and, more recently (1998-2005), has shown evidence of a declining trend of approximately 22.3% (FFWCC 2007a; FFWCC 2007b; Witherington et al. 2009). This is likely due to a decline in the number of nesting females within the population (Witherington et al. 2009). Nesting data from the Archie Carr Refuge (one of the most important nesting locations in southeast Florida) over the last 6 years shows nests declined from approximately 17,629 in 1998 to 7,599 in 2004, also suggesting a decrease in population size[1]. Loggerhead nesting is thought to consist of just 60 nesting females in the Caribbean and Gulf of Mexico (NMFS 2006f). Based upon the small sizes of almost all nesting aggregations in the Atlantic, the large numbers of individuals killed in fisheries, and the decline of the only large nesting aggregation, we suspect that the extinction probabilities of loggerhead sea turtle populations in the Atlantic are only slightly lower than those of populations in the Pacific.

The northwestern Atlantic DPS is considered to have a high risk of quasi-extinction in the foreseeable future and has one of the highest potentials for future decline of all loggerhead DPSs (Conant et al. 2009). All "subpopulations" showed some degree of decline.

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 Mediterannean Sea, Anthorpogenic threats appear

1 While this is a long period of decline relative to the past observed nesting pattern at this location, aberrant ocean surface temperatures complicate the analysis and interpretation of these data. Although caution is warranted in interpreting the decreasing nesting trend given inherent annual fluctuations in nesting and the short time period over which the decline has been noted, the recent nesting decline at this nesting beach is reason for concern.

to disproportionally impact larger (more fecund) loggerheads (Bellido et al. 2010).

The major factors inhibiting their recovery include mortalities caused by fishery interactions and degradation of the beaches on which they nest. Shrimp trawl fisheries account for the highest number of captured and killed loggerhead sea turtles. Along the Atlantic coast of the U.S., the NMFS estimated that shrimp trawls capture almost 163,000 loggerhead sea turtles each year in the Gulf of Mexico, of which 3,948 die. Each year, various fisheries capture about 2,000 loggerhead sea turtles in Pamlico Sound, of which almost 700 die.

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).

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. 2008). 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 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).

The loggerhead BRT identified bycatch as the primary threat to northwestern DPS loggerheads; boat strikes and marine debris are considered additional significant threats (Conant et al. 2009). Other threats summarized above are also issues for the survival and recovery of the northwestern Atlantic DPS.

Critical Habitat. The NMFS has not designated critical habitat for loggerhead 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 species in the action area.

Physical and Oceanographic Features

The presence of key habitat features, such as shelter or foraging opportunities, are the primary reasons why listed individuals occur where they do. In the marine environment, this is fundamentally built upon local physical and oceanographic features that influence the marine environment. As such, we describe the physical and oceanographic environment here to establish a rationale for why listed species occur in the action area at the levels we observe or expect.

The Gulf of Mexico generally lacks many of the features, such as seamounts, trenches, and other sharp bathymetric relief, that in other oceanic areas enhance primary and secondary marine productivity exploited by listed marine species (Brooks 1973). Exceptions to the area the DeSoto, Mississippi, Alaminos, and Keathley Canyons; all are located within the general region of the action area and, at least the DeSoto Canyon, have been linked with sperm whale foraging.

Although bathymetrically-generated productivity enhancement is generally lacking, foraging opportunities are none the less present and mostly predicated upon oceanographic features within the basin. As with most other marine oceanic regions, the Gulf of Mexico consists of surface, intermediate, and deep water layers separated by thermal- and saline-based water densities. Deeper waters tend to contain higher nutrient levels, but are unavailable for photosynthetic organisms due to lack of light at the depths these layers exist. However, oceanographic features can bring this deeper water to or near the surface, allowing for enhanced productivity in these

areas.

In the action area, the Loop Current (the dominant surface current in the Gulf of Mexico) is the primary feature leading to enhanced productivity. The flow of the current varies within and between years, but enters the Gulf of Mexico through the Yucatan Channel and exits through the Florida Straits. As the Loop Current enters the northern Gulf of Mexico, its track becomes unstable, producing large anticyclonic (warm-core) eddies (Biggs 1992). These warm-core eddies then move west or west-southwest until they encroach onto the continental shelf after several months and disperse (Maul and Vukovich 1993; Wiseman and Sturges 1999). However, during their existence, small cold-core eddies form on the periphery of the warm-core eddies. These cold-core eddies produce upwelling, bringing nutrient-rich water to the surface. In addition, streams of warm water flow countercurrent but alongside the Loop Current, producing upwelling along the current's periphery. The flow of the Mississippi River also introduces enhanced nutrient waters in the northern Gulf of Mexico. Chlorophyll α concentrations are particularly high near the river's mouth (Scott et al. 2001).

The predominant phytoplankton groups in the Gulf of Mexico include blue-green algae, pelagophytes, prymnesiophytes, and prochlorophytes (Jochens et al. 2002). The later three dominate most seasons on the outer continental shelf and slope (including the action area). Phytoplankton levels in the Gulf of Mexico are generally low, although hot spots do occur, such as in association with aforementioned features (Biggs 1992; Biggs and Müller-Karger 1994; Biggs and Ressler 2001). Zooplankton, which feed upon phytoplankton, are also concentrated in these areas (Biggs and Ressler 2001). Euphausiids appear to be particularly numerous in these features (Biggs et al. 1997).

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 continue 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 effects 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 reproduction, the distribution and abundance of prey and abundance of competitors or predators. For species that undergo long migrations, individual movements are usually associated with 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). For sea turtles, warming ocean temperatures may extend poleward the habitat which they can utilize (Poloczanska et al. 2009). However, seagrass habitats have declined by 29% in the last 130 years and 19% of coral reefs have been lost due to human degradation, reducing lower latitude habitat for some sea turtle species (Poloczanska et al. 2009). Primary production is estimated to have declined by 6% between the early 1980s and 2010, making foraging more difficult for marine species (Hoegh-Guldberg and Bruno 2010).

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. 2009a). 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).

Natural variations in climactic patterns also have impacts on the Gulf of Mexico, particularly El Niño-Southern Oscillation and the North Atlantic Oscillation (NAO). Hurricanes tend to be stronger and more frequent during La Niña years than El Niño years (Twilley et al. 2001). Similarly, a positive NAO index years are associated with more frequent hurricanes in the Gulf of Mexico (Elsner et al. 2000).

Entrapment and Entanglement in Fishing Gear

Fishery interaction remains a major factor in sea turtle recovery and, frequently, the lack thereof. NMFS (2002) estimated that 62,000 loggerhead sea turtles have been killed as a result of incidental capture and drowning in shrimp trawl gear. Although turtle excluder devices and other bycatch reduction devices have significantly reduced the level of bycatch to sea turtles and other marine species in U.S. waters, mortality still occurs in Gulf of Mexico waters.

In addition to commercial bycatch, recreational hook-and-line interaction also occurs. Cannon and Flanagan (1996) reported that from 1993 to 1995, at least 170 Kemp's ridley sea turtles were hooked or tangled by recreational hook-and-line gear in the northern Gulf of Mexico. Of these, 18 were dead stranded turtles, 51 were rehabilitated turtles, five died during rehabilitation, and 96 were reported as released by fishermen.

Commercial and Subsistence Harvest

Three of the biggest threats to sea turtles result from harvest for commercial and subsistence use. These include egg harvest, the harvest of females on nesting beaches, and directed hunting of sea turtles in foraging areas. These factors have led to the precipitous declines in worldwide sea turtle populations. In the Atlantic, green sea turtles are captured and killed in turtle fisheries in Colombia, Grenada, the Lesser Antilles, Nicaragua, St. Vincent and the Grenadines; the turtle fishery along the Caribbean coast of Nicaragua, by itself, has captured more than 11,000 green sea turtles annually over the past decade (Bräutigam and Eckert 2006a; Lagueux 1998). While these threats have been largely eliminated in Florida due to successful conservation measures, the hunting of juvenile and adult turtles continues both legally and illegally in many foraging areas where green sea turtles originating from Florida are known to occur (Chacon 2002; Fleming 2001). The killing of nesting hawksbill females continues to threaten the stability of hawksbill subpopulations in many areas. The centuries-old historic trade in tortoise shell greatly impacted hawksbill populations in the Insular Caribbean. Increases in nesting hawksbills in the region coincide with the decline of international trade in hawksbill shell (Milliken and Tokunaga 1987), and in particular with the 90% reduction in the annual take of large hawksbills from Cuban waters (Carrillo et al. 1999).

Coastal Development

A large number of coastal development activities can have adverse impacts on sea turtles. Most of these, such as construction of breakwaters and seawalls, lighting, building construction, and vehicular beach traffic are significant mostly or only for nesting sea turtles and their hatchlings. The action area does generally not host sea turtle nesting (or where it does, the establishment of state and national refuges limit or prevent human development), we focused on impacts to sea turtles in the marine environment, namely dredging activity. Dredging has been documented to capture or kill 168 sea turtles from 1995 to 2009 in the Gulf of Mexico, including 97 loggerheads, 35 Kemp's ridleys, 32 greens, and three unidentified sea turtles (USACOE 2010).

Oil and Gas Development

The northern Gulf of Mexico is the location of massive industrial activity associated with oil and gas extraction and processing. Over 4,000 oil and gas structures are located outside of state waters in the northern Gulf of Mexico; 90% of these occur off Louisiana and Texas (USN 2009). This is both detrimental and beneficial for sea turtles. These structures appreciably increase the amount of hard substrate in the marine environment, providing shelter and foraging opportunities for species like loggerhead sea turtles (Parker et al. 1983; Stanley and Wilson 2003). However, the Minerals Management Service requires that structures must be removed within one year of lease termination. Many of these structures are removed by explosively severing the underwater supportive elements, which produces a shock wave that kills, injures, or disrupts marine life in the blast radius (Gitschlag et al. 1997). For sea turtles, this means death or serious injury for individuals within a few hundred meters of the structure and overt behavioral (potentially physiological) impacts for individuals further out (Duronslet et al. 1986; Klima et al. 1988). Although observers and procedures are in place to mitigate impacts to sea turtles (i.e., not blasting when sea turtles are present), not all sea turtles are observed all the time and low-level sea turtle injury and mortality still occurs (Gitschlag and Herczeg 1994; Gitschlag et al. 1997); two loggerheads were killed in August 2010 (G. Gitschlagg, NOAA, pers. comm.). Current annual authorized takes due to MMS' OCS oil and gas exploration, development, production, and abandonment activities are 30 sea turtles, including no more than one each of Kemp's ridley, green, hawksbill, or leatherback turtles and no more than ten loggerhead turtles (NMFS 1988).

Pollution

The Gulf of Mexico is a sink for massive levels of pollution from a variety of marine and terrestrial sources, which ultimately can interfere with ecosystem health and particularly that of sea turtles (see *Status of Listed Resources* section). Sources include the petrochemical industry in and along the Gulf of Mexico, wastewater treatment plants, septic systems, industrial facilities, agriculture, animal feeding operations, and improper refuse disposal. The Mississippi River drains 80% of United States cropland (including the fertilizers, pesticides, herbicides, and other contaminants that are applied to it) and discharges into the Gulf of Mexico near the action area (MMS 1998). Agricultural discharges, as well as discharges from large urban centers (ex.: Houston and New Orleans) contribute contaminants as well as coliform bacteria to Gulf of Mexico habitats (Garbarino et al. 1995). These contaminants can be carried long distances from terrestrial or nearshore sources and ultimately accumulate in offshore pelagic environments (USCOP 2004). The ultimate impacts of this pollution are poorly understood.

Significant attention has been paid to nutrient enrichment of Gulf of Mexico waters, which leads to algal blooms (including harmful algal blooms), oxygen depletion, loss of seagrass and coral reef habitat, and the formation of a hypoxic "dead zone" (USCOP 2004). This hypoxic event occurs annually from as early as February to as late as October, spanning roughly 12,700 km^2 (although in 2005 the "dead zone" grew to a record size of 22,000 km^2) from the Mississippi River Delta to Galveston, Texas (LUMCON 2005; MMS 1998; Rabalais et al. 2002; USGS 2010). Although sea turtles do not extract oxygen from sea water, numerous staple prey items of sea turtles, such as fish, shrimp, and crabs, do and are killed by the hypoxic conditions (Craig et al. 2001). More generally, the "dead zone" decreases biodiversity, alters marine food webs, and destroys habitat (Craig et al. 2001; Rabalais et al. 2002). High nitrogen loads entering the Gulf of Mexico from the Mississippi River is the likely culprit; nitrogen concentrations entering the

Gulf of Mexico have increased three fold over the past 60 years (Rabalais et al. 2002).

Marine debris is also a significant problem for the health of Gulf of Mexico ecosystems. For sea turtles, marine debris is a problem due primarily to individuals ingesting debris and blocking the digestive tract, causing death or serious injury (Laist et al. 1999; Lutcavage et al. 1997a). Gulko and Eckert (2003) estimated that between one-third and one-half of all sea turtles ingest plastic at some point in their lives; this figure is supported by data from Lazar and Gracan (Lazar and Gračan 2010), who found 35% of loggerheads had plastic in their gut. A Brazilian study found that 60% of stranded green sea turtles had ingested marine debris (primarily plastic and oil; (Bugoni et al. 2001)). Loggerhead sea turtles had a lesser frequency of marine debris ingestion. Plastic is possibly ingested out of curiosity or due to confusion with prey items; for example, plastic bags can resemble jellyfish (Milton and Lutz 2003). Marine debris consumption has been shown to depress growth rates in post-hatchling loggerhead sea turtles, elongating the time required to reach sexual maturity and increasing predation risk (McCauley and Bjorndal 1999). Studies of shore cleanups have found that marine debris washing up along the northern Gulf of Mexico shoreline amounts to about 100 kg/km (ACC 2010; LADEQ 2010; MASGC 2010; TGLO 2010). Sea turtles can also become entangled and die in marine debris, such as discarded nets and monofilament line (Laist et al. 1999; Lutcavage et al. 1997a; NRC 1990a; O'Hara et al. 1988).

Oil Spills and Releases

Oil pollution has been a significant concern in the Gulf of Mexico for several decades due to the large amount of extraction and refining activity in the region. Routine discharges into the northern Gulf of Mexico (not including oil spills) include roughly 88,200 barrels of petroleum per year from municipal and industrial wastewater treatment plants and roughly 19,250 barrels from produced water discharged overboard during oil and gas operations (MMS 2007b; USN 2008). These sources amount to over 100,000 barrels of petroleum discharged into the northern Gulf of Mexico annually. Although this is only 10% of the amount discharged in a major oil spill, such as the Exxon *Valdez* spill (roughly 1 million barrels), this represents a significant and "unseen" threat to Gulf of Mexico wildlife and habitats. Furthermore, accidental oil spills may amount to less than 24,000 barrels of oil discharged annually in the northern Gulf of Mexico, making non-spilled oil normally one of the leading sources of oil discharge into the Gulf of Mexico (MMS 2007a). The other major source from year to year is oil naturally seeping into the northern Gulf of Mexico. Although exact figures are unknown, natural seapage is estimated at between 120,000 and 980,000 barrels of oil annually (MacDonald et al. 1993; MMS 2007b).

Although non-spilled oil is the primary contributor to oil introduced into the Gulf of Mexico, concern over accidental oil spills is well-founded. Over five million barrels of oil and one million barrels of refined petroleum products are transported in the northern Gulf of Mexico daily (MMS 2007b); worldwide, it is estimated that 900,000 barrels of oil are released into the environment as a result of oil and gas activities (Epstein and (Eds.). 2002). Even if a small fraction of the annual oil and gas extraction is released into the marine environment, major, concentrated releases can result in significant environmental impacts. Due to the density of oil extraction, transport, and refining facilities in the Houston/Galveston and Mississippi Delta areas (and the extensive activities taking place at these facilities), these locations have the greatest probability of experiencing oil spills. Oil released into the marine environment contains aromatic organic chemicals known to be toxic to a variety of marine life; these chemicals tend to

dissolve into the air to a greater or lesser extent, depending upon oil type and composition (Yender et al. 2002). Solubility of toxic components is generally low, but does vary and can be relatively high (0.5-167 parts per billion; (Yender et al. 2002)). Use of dispersants can increase oil dispersion, raising the levels of toxic constituents in the water column, but speeding chemical degradation overall (Yender et al. 2002). The remaining oil becomes tar, which forms floating balls that can be transported thousands of kilometers into the North Atlantic. The most toxic chemicals associated with oil can enter marine food chains and bioaccumulate in invertebrates such as crabs and shrimp to a small degree (prey of some sea turtles (Law and Hellou 1999; Marsh et al. 1992)), but generally do not bioaccumulate or biomagnify in finfish (Baussant et al. 2001; Meador et al. 1995; Varanasi et al. 1989; Yender et al. 2002). Sea turtles are known to ingest and attempt to ingest tar balls, which can cause their jaws to become adhered or block their digestive systems, impairing foraging or digestion and potentially causing death (NOAA 2003). Dispersants reduce the formation of tar balls. Although the effects of dispersant chemicals on sea turtles is unknown, testing on other organisms have found currently used dispersants to be less toxic than those used in the past (NOAA 2003). It is possible that dispersants can interfere with surfactants in the lungs (surfactants prevent the small spaces in the lungs from adhering together due to surface tension, facilitating large surface areas for gas exchange), as well as interfere with digestion, excretion, and salt gland function (NOAA 2003). Oil exposure can also cause acute damage upon direct exposure to oil, including skin, eye, and respiratory irritation, reduced respiration, burns to mucous membranes such as the mouth and eyes, diarrhea, gastrointestinal ulcers and bleeding, poor digestion, anemia, reduced immune response, damage to kidneys or liver, cessation of salt gland function, reproductive failure, and death (NOAA 2003; NOAA 2010b; Vargo et al. 1986b; Vargo et al. 1986a; Vargo et al. 1986c). Nearshore spills or large offshore spills can oil beaches on which sea turtles lay their eggs, causing birth defects or mortality in the nests (NOAA 2003; NOAA 2010b).

Oil can also cause indirect effects to sea turtles through impacts to habitat and prey organisms. Seagrass beds may be particularly susceptible to oiling as oil contacts grass blades and sticks to them, hampering photosynthesis and gas exchange (Wolfe et al. 1988). If spill cleanup is attempted, mechanical damage to seagrass can result in further injury and long-term scarring. Loss of seagrass due to oiling would be important to green sea turtles, as this is a significant component of their diets (NOAA 2003). The loss of invertebrate communities due to oiling or oil toxicity would also decrease prey availability for hawksbill, Kemp's ridley, and loggerhead sea turtles (NOAA 2003). Furthermore, Kemp's ridley and loggerhead sea turtles, which commonly forage on crustaceans and mollusks, may ingest large amounts of oil due oil adhering to the shells of these prey and the tendency for these organisms to bioaccumulate toxins found in oil (NOAA 2003). It is suspected that oil adversely impacted the symbiotic bacteria in the gut of herbivorous marine iguanas when the Galapagos Islands experienced an oil spill, contributing to a >60% decline in local populations the following year. The potential exists for green sea turtles to experience similar impacts, as they also harbor symbiotic bacteria to aid in their digestion of plant material (NOAA 2003). Dispersants are believed to be as toxic to marine organisms as oil itself.

Several oil spills have impacted the northern Gulf of Mexico over the past few years, largely due to hurricanes. The impacts of Hurricane Ivan in 2004 on the Gulf Coast included pipeline damage causing 16,000 barrels of oil to be released and roughly 4,500 barrels of petroleum products from other sources (BOEMRE 2010; USN 2008). The next year, Hurricane Katrina caused widespread damage to onshore oil storage facilities, releasing 191,000 barrels of oil

(LHR 2010). Another 4,530 barrels of oil were released from 70 other smaller spills associated with hurricane damage. Shortly thereafter, Hurricane Rita damaged offshore facilities resulting in 8,429 barrels of oil to be released (USN 2008).

Major oil spills have impacted the Gulf of Mexico for decades (NMFS 2010). Until 2010, the largest oil spill in North America occurred in the Bay of Campeche (1979), when a well "blew out", allowing oil to flow into the marine environment for nine months, releasing 2.8-7.5 million barrels of oil. Oil from this release eventually reached the Texas coast, including the Kemp's ridley sea turtle nesting beach at Rancho Nuevo, where 9,000 hatchlings were airlifted and released offshore (NOAA 2003). Over 7,600 m^3 of oiled sand was eventually removed from Texas beaches and 200 gallons of oil were removed from the area around Rancho Nuevo (NOAA 2003). Eight dead and five live sea turtles were recovered during the oil spill event; although cause of deaths were not determined, oiling was suspected to play a part (NOAA 2003). Also in 1979, the oil tanker *Burmah Agate* collided with another vessel near Galveston, Texas, causing an oil spill and fire that ultimately released 65,000 barrels of oil into estuaries, beachfronts, and marshland along the northern and central Texas coastline (NMFS 2010). Clean-up of these areas was not attempted due to the environmental damage such efforts would have caused. Another 195,000 barrels of oil are estimated to have been burned in a multi-month-long fire aboard the *Burmah Agate* (NMFS 2010). The tanker *Alvenus* grounded in 1984 near Cameron, Louisiana, spilling 65,500 barrels of oil which spread west along the shoreline to Galveston (NMFS 2010). One oiled sea turtle was recovered and released (NOAA 2003). In 1990, the oil tanker *Megaborg* experienced an accident near Galveston during the lightering process and released 127,500 barrels of oil, most of which burned off in the ensuing fire (NMFS 2010).

On April 20, 2010, a fire and explosion occurred aboard the semisubmersible drilling platform *Deepwater Horizon* roughly 80 km southeast of the Mississippi Delta (NOAA 2010a). The platform had 17,500 barrels of fuel aboard, which likely burned, escaped, or sank with the platform (NOAA 2010a). However, once the platform sank, the riser pipe connecting the platform to the wellhead on the seafloor broke in multiple locations, initiating an uncontrolled release of oil from the exploratory well. Over the next three months, oil was released into the Gulf of Mexico, resulting in oiled regions of Texas, Louisiana, Mississippi, Alabama, and Florida and widespread oil slicks throughout the northern Gulf of Mexico that closed more than one-third of the Gulf of Mexico Exclusive Economic Zone to fishing due to contamination concerns. Apart from the widespread surface slick, massive undersea oil plumes formed, possibly through the widespread use of dispersants and reports of tarballs washing ashore throughout the region were common. Although estimates vary, NOAA has estimated that 4.9 million barrels of oil were released (Lubchenco et al. 2010). A total of 720 sea turtles have been verified in the spill zone of which 172 were verified as having oil exposure (NOAA 2010c). However, specific causes of injury or death have not yet been established for many of these individuals as investigations into the role of oil in these animals' health status continue.

Hurricanes

The Gulf of Mexico is prone to major tropical weather systems, including tropical storms and hurricanes. The impacts of these storms on sea turtles in the marine environment is not known, but storms can cause major impacts to sea turtle eggs on land, as nesting frequently overlaps with hurricane season, particularly Kemp's ridley sea turtles (NRC 1990a). Mortality can result both

from drowning of individuals while still in the egg or emerging from the nest as well as causing major topographic alteration to beaches, preventing hatchling entry to marine waters. Kemp's ridley sea turtles are likely highly sensitive to hurricane impacts, as their only nesting locations are in a limited geographic area along southern Texas and northern Mexico (Milton et al. 1994). In 2010, Hurricane Alex made landfall in this area; surprisingly, few nests were lost (Jaime Pena, Gladys Porter Zoo, pers. comm.). Tropical storm Hermine arrived too late to impact eggs or hatchlings at Rancho Nuevo (Donna Shaver, NPS, pers. comm.).

Vessel Strike

Vessel strike of sea turtles is a poorly-studied threat to sea turtles, but has the potential to be highly-significant (Work et al. 2010). All sea turtles must surface to breath and several species are known to bask at the surface for long periods, including loggerhead sea turtles. Although sea turtles can move rapidly, sea turtles apparently are not well able to move out of the way of vessels moving at more than 4 km/hr; most vessels move far faster than this in open water (Hazel et al. 2007; Work et al. 2010). This, combined with the massive level of vessel traffic in the Gulf of Mexico, has the potential to result in frequent injury and mortality to sea turtles in the region (MMS 2007b). Hazel et al. (2007) suggested that green sea turtles may use auditory cues to react to approaching vessels rather than visual cues, making them more susceptible to strike as vessel speed increases. Each state along the Gulf of Mexico has several hundred thousand recreational vessels registered, including Florida with nearly one million; the highest number of registered boats in the United States, and Texas with over 600,000 (ranked 6th nationally)(NMMA 2007; USCG 2003; USCG 2005). Commercial vessel operations are also extensive. Vessels servicing the offshore oil and gas industry are estimated to make 115,675-147,175 trips annually, apart from commercial vessels travelling to and from some of the largest ports in the United States (such as New Orleans and Houston)(MMS 2007a; USN 2008).

Sea turtles may also be harassed by the high level of helicopter activity over Gulf of Mexico waters. It is estimated that between roughly 900,000 and 1.5 million helicopter take-offs and landings are undertaken in association with oil and gas activities in the Gulf of Mexico annually (NRC 1990a; USN 2008). This likely includes numerous overflights of sea turtles; an activity which has been observed to startle and at least temporarily displace sea turtles (USN 2009).

Naval activities

Naval activity, notable sonar use during training exercises, has gained notoriety for its coincidence with marine mammal strandings. However, other activities (also during training exercises in designated naval operating areas and training ranges) also have the potential to adversely impact marine mammals and sea turtles. Species occurring in the action area could experience stressors from several naval training ranges or facilities listed below. Listed individuals travel widely in the North Atlantic and could be exposed to naval activities in several ranges.

- Northeast Training Range Complex, where leatherback sea turtles forage,
- The Virginia Capes, Cherry Point, and Jacksonville-Charleston Operating Areas, which are situated consecutively along the migratory corridor for sea turtles, and
- The Key West, Gulf of Mexico, Bermuda, and Puerto Rican Complexes have the potential to overlap the distribution of sperm whales and sea turtles species.

Naval activities to which individuals could be exposed include, among others, vessel and aircraft transects, munition detonations, and sonar use. Responses by marine mammals could include no response, short-term and long-term behavioral responses and changes (altered vocal activity, changes in swimming speed and direction, respiration rates, dive times, and social interactions), temporary or permanent hearing loss, debris ingestion, ship-strike injury, and death.

Although naval vessels represent a small fraction of the total sound level and are designed to operate quietly, these ships are large and equipped with high-output sonar equipment such as ANISQS-53C tactical sonar, which produces signals at source levels of 235 dB re 1 μPa_{rms} at 1 m. The signals emitted from these devices have the potential to affect marine mammals in the action area; however, empirical data are limited.

Scientific and Research Activities

Scientific research permits issued by the NMFS currently authorize studies listed species in the Atlantic Ocean, which may extend into portions of the action area for the proposed actions. Authorized research on ESA-listed sea turtles includes capture, handling, and restraint, satellite, sonic, and PIT tagging, blood and tissue collection, lavage, ultrasound, captive experiments, laparoscopy, and imaging. Research activities involve “takes” by harassment, with some resulting mortality. Additional “take” is likely to be authorized in the future as additional permits are issued. It is noteworthy that although the numbers tabulated below represent the maximum number of “takes” authorized in a given year, monitoring and reporting indicate that the actual numbers of “takes” rarely approach the number authorized. Therefore, it is unlikely that this level of exposure indicated below has or will occur in the near term. However, our analysis assumes that these “takes” will occur since they have been authorized. It is also noteworthy that these “takes” are distributed across the Atlantic Ocean, mostly from Florida to Maine, and in the eastern Gulf of Mexico. Although sea turtles are generally wide-ranging, we do not expect many of the authorized “takes” to involve individuals who would also be “taken” under the proposed research.

Tables 4-7 describe the cumulative number of takes for each listed species in the action area authorized in scientific research permits

Table 4. Sperm whale takes in the North Atlantic Ocean.

Year	Approach	Biopsy	Suction cup tagging	Acoustic playback
2009	5,560	375	820	920
2010	4,110	400	520	120
2011	4,010	400	520	120
2012	1,950	125	10	0
2013	1,865	100	10	0
2014	865	0	0	0
Total	18,360	1,400	1,880	1,160

Permit numbers: 633-1778, 775-1875, 909-1719, 948-1692, 981-1707, 1036-1744, 1121-1900, 10014, 14451, 14586, 15575, 16109, and 16473

Table 5. Leatherback sea turtle takes in the North Atlantic Ocean.

Year	Capture/handling/restraint	Satellite, sonic, or pit tagging	Blood/tissue collection	Lavage	Ultrasound	Mortality
2009	1,357	1,357	1,331	197	188	2
2010	1,421	1,421	1,394	197	188	1
2011	1,709	1,709	1,682	197	189	3.4
2012	736	736	709	187	189	2.6
2013	696	696	669	187	189	0.6
2014	507	507	481	10	1	0.6
Total	6,426	6,426	6,266	975	944	10.2

Permit numbers: 1506, 1527, 1540, 1544, 1551, 1552, 1557, 1570, 1571, 1576, 10014, 13543, 14506, 14586, 14655, 14726, 15112, 15552, 15556, 15575, 15672, 15802, 16109, 16194, and 16253.

Table 6. Loggerhead sea turtle takes in the North Atlantic Ocean.

Year	Capture/handling /restraint	Satellite, sonic, or pit tagging	Blood/tissue collection	Lavage	Ultrasound	Captive experiment	Laparoscopy	Imaging	Mortality
2009	5,462	5,462	5,044	1,165	1,322	200	109	123	111
2010	5,464	5,464	5,046	1,205	1,322	200	109	116	111
2011	7,165	7,165	6,097	1,420	1,667	200	148	114	122.2
2012	4,791	4,791	3,741	1,370	1,429	200	161	114	29.8
2013	4,363	4,363	3,313	1,369	1,429	200	161	114	22.6
2014	3,211	3,211	2,166	255	453	0	52	0	22.6
Total	30,456	30,456	25,407	6,784	7,169	1,000	740	581	419.2

Permit numbers: 1450, 1462, 1501, 1506, 1507, 1522, 1526, 1527, 1540, 1544, 1551, 1552, 1570, 1571, 1576, 1599, 10014, 10022, 13306, 13307, 13543, 13544, 14249, 14622, 14506, 14508, 14655, 14726, 15112, 15552,15566, 15575, 15606, 15802,16146, 16253, and 16598.

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 seismic survey by the USGS and issuance of the IHA by the NMFS' Permits Division for "takes" of marine mammals during the seismic studies would expose listed species to seismic airgun pulses as well as other stressors. 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 and physiological 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 acoustic "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

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)

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. vessel strikes
4. acoustic interference from engine noise
5. entanglement in towed hydrophone streamers and other gear
6. sound fields produced by airguns

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 is 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 *Pelican* 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.

Although noise originating from vessel propulsion will propagate into the marine environment, this amount would be so small as to be discountable. The *Pelican'*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 *Pelican* will be traveling at generally slow speeds, reducing the amount of noise produced by the propulsion system and the probability of a vessel strike (Kite-Powell et al. 2007; Vanderlaan and Taggart 2007). Our expectation of vessel strike is discountably small due to the roughly 100,000 miles the *Pelican* has travelled without a strike and general expected movement of marine mammals and sea turtles away or parallel to the *Pelican* (Hauser and Holst 2009; Holst 2009; Holst 2010; Holst and Smultea 2008a). Speed will also be very slow in general, a factor that has been found to greatly decrease the probability of vessel strike. All things considered, we feel the potential for vessel strike or acoustic interference from propulsion and machinery noise is discountable.

The towed hydrophone streamer 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). Gear is regularly deployed from research vessels similar to the *Pelican* (amounting to tens if not hundreds of thousands of deployments or more), but we could not identify a case of entanglement for a large whale or sea turtle (other than the case noted) for the *Pelican* or any other similar research vessel. Although the towed hydrophone streamer and other deployed gear could come in direct contact with a listed species, entanglements are highly unlikely and considered discountable based upon investigation into the use of these devices during the activities of other oceanographic activities.

Accordingly, this consultation focused on acoustic energy stressors that are likely to be generated by the airgun array during proposed seismic activities which may adversely affect ESA-listed species.

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. These thresholds are in the final stages of being updated and, once evualated with internal external peer review as well as public comment, will be a basis of evaluating going forward. Until this time, older but stringently developed criteria are applied in conjunction with additional best available science.

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 USGS and NMFS' Permits Division provided density estimates for listed whales in the action area. Review of the local survey data and surveys in the wider Gulf of Mexico 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 NOAA from 1994-2006 during seasonal marine mammals surveys through the Gulf of Mexico. Data from these surveys were combined, statistically assessed for use in developing density estimates for U.S. Navy ranges in the region, and published in USN (USN 2007). These densities were further decomposed by area by the Bureau of Ocean Energy Management for use in estimating impacts of oil and gas activities in the Gulf of Mexico (BOEM 2012).

Sperm whales, leatherback sea turtles, and loggerhead sea turtles are expected to be abundant in the action area as a consequence of the upwelling features that frequently occur in offshore waters of the Gulf of Mexico. These features are expected to enhance local primary and secondary productivity, which would provide enhanced foraging opportunities for these species.

USGS also estimated the exposure radii around the proposed *Pelican* 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 the two-airgun array used during the proposed study are provided in Table 1 on page 4. The maximum distance from airguns where received levels might reach 160 dB re 1 μPa_{rms} (i.e., from the full two-gun array) at 2,000 m depth (maximum depth at which listed species are expected to occur) is 670 m with a three meter tow depth. A thorough review of available literature (see *Response Analysis*) supports this level as a general point at which whales tend to show some avoidance response to received seismic sound. The USGS' 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 exposures 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. 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 USGS 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, 180, and 190 dB re 1 μPa_{rms} propagation distances are well within the visual range of the *Pelican* and its observers, it is unlikely that enough observers will be present to effectively scan the area. Shut-down procedures are unlikely to be completely effective at eliminating the co-occurrence of listed individuals within the sound field $\geq$160 dB re 1 μPa_{rms}.

Marine Mammals

Exposure of Listed Mammals to Airguns

Exposure estimates stem from the best available information on whale densities and a planned ensonified area of approximately 2,479 km^2 along survey track lines, including areas of repeated exposure from adjacent track lines and an increase in survey tracklines of 25% to account for turns, equipment testing, and turns.

Our exposure estimates were calculated by using the density per 1,000 km^2 (4.90 for sperm whales) multiplied by the total survey track area (2,479 km^2) to obtain the total number of potential exposures (13, rounded up to the next whole number). This number is not larger than the average sperm whale group size in the northern Gulf of Mexico (6.9 individuals) and was not increased to account for group size being larger than the estimate (Jaquet and Gendron 2009b).

The USGS originally calculated re-exposure based upon area of exposure with and without re-exposure, which was determined to be between 5.3 and 6.5 fold greater for the area of re-exposure. This, combined with an assumption that an exposed individual would not shift location during the duration of the seismic survey, was used to determine that a given individual would be exposed between five and seven times each. We explored alternative ways to estimate the level of re-exposure that individuals would likely experience. 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 with which we can accurately account for the probability and effect 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. We can, however, estimate a range of exposures. It is reasonable to expect, based upon review of observed effects of seismic sound exposure to marine mammals, that some individuals will move a distance of up to several kilometers away due to individual or situational circumstances. As such, it is reasonable to expect that some individuals will receive a single exposure and vacate the action area. Other individuals may move, but move to locations where re-exposure would occur, either due to the direction or short distance they travel. As such, it is reasonable that some individuals may be exposed up to the maximum number of transect lines that each box would be surveyed (152) (statistically, this is unlikely unless the stressor does not represent a significant one motivating the individual to vacate the area, the motivation to stay in a specific area is high, or the stressor acts as a direct or indirect attractive feature). This is particularly significant given that marine mammals in general and sperm whales in the Gulf of Mexico in particular tend to return to specific areas for foraging and breeding, or use particular migratory corridors. It is possible, but not reasonable, that an individual may be ensonified up to 152 times (maximum number of re-exposures of a given location in each of the two survey locations). We expect the maximum number of exposures an individual would receive to be significantly less than this, although we are unable to articulate precisely how much less. We expect up to 13 individuals may be exposed. This represents up to 1.7% of the local population (763 individuals, based upon recent estimates) (NMFS 2012).

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, sperm whales in the area likely consist of groups of adult females and their offspring and generally consist of more females than males in the group. Therefore, we expect a female bias to sperm whale exposure. Exposure to adult males is expected to be much lower than to other age and sex class combinations.

Sea Turtles

Exposure of Listed Turtles to Airguns

The USGS did not quantify the number of exposure events or the number of exposures per individual sea turtle. Leatherback and loggerhead 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. These factors include high-quality foraging habitat as well as our best understanding of species movement patterns.

The action area includes a region where periodic current features provide enhanced foraging opportunities. As with listed whales, this area has the potential to draw large numbers of sea turtles (particularly leatherback and loggerhead). 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. Leatherbacks and loggerheads have been documented regularly in offshore Gulf of Mexico waters. We expect leatherback and loggerhead sea turtles will be exposed to airgun discharges, with exposure being higher for females than for males due to the sex bias present in known populations. No hatchlings of any species are expected to be exposed (we are unaware of hatchlings occurring in the region), although it is possible that all other life stages may be exposed. We are not able to quantify the number of exposures individuals of each species would

experience, but based upon the widely dispersed and far-ranging nature of these species, we expect that a small percentage of the DPS or population as a whole would potentially be exposed.

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 or within 1.8 km during full airgun operations, 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. 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 *Pelican's* 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).

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. 1995a). The *Pelican's* airguns will emit a 0.1 s pulse when fired every 6-10 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.

Although seismic sound pulses begin as short, discrete sounds, they interact with the marine environment and lengthen through processes such as reverberation. This means that in some cases, such as shallow water environments, seismic sound can become part of the acoustic background. Few studies of how impulsive sound in the marine environment deforms from short bursts to lengthened waveforms exist, but can apparently add significantly to acoustic background (Guerra et al. 2011), potentially interfering with the ability of animals to hear otherwise detectible sounds in their environment.

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. 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). 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). 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 $\mu Pa_{p\text{-}p}$ (Madsen et al. 2002; McCall Howard 1999). Some exposed individuals may cease calling in response to the *Pelican's* airguns. If individuals ceased calling in response to the *Pelican's* airguns during the course of the proposed survey, the effect would likely be temporary.

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. 2000c; Madsen et al. 2006; Miller et al. 2009; Moulton et al. 2006; 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 $\mu Pa_{p\text{–}p}$, although other behavioral reactions were not noted by several authors (Gordon et al. 2006; 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 2003; 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. 1995b). 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). Noteworthy is the timing of the proposed seismic survey, which will overlap the breeding season of sperm whales n the North Atlantic {NMFS, 2010 #151142}. It is not known whether the local population will be engaged in reproductive behavior at the time of the survey. However, if individuals are, we expect individuals to ignore stressors associated with the proposed survey and continue baseline activtities or move a short distance away and re-engage in breeding behavior.

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. 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 *Pelican*'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 (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 μPa · 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. 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. 1996a; 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; Dalen and Knutsen 1986). 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 $\mu Pa_{p\text{-}p}$. Exposure to airguns at close range were found to produce balance issues in exposed fry (Dalen and Knutsen 1986). Exposure of monkfish and capelin eggs at close range to airguns did not produce differences in mortality compared to control groups (Payne et al. 2009).

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 $\mu Pa_{0\text{-}p}$ and alarm responses at >177 dB re 1 $\mu Pa_{0\text{-}p}$ (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 $\mu Pa_{0\text{-}p}$. 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 $\mu Pa_{0\text{-}p}$ 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. 1996a). 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 $\mu Pa_{0\text{-}p}$, 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 $\mu Pa_{p\text{-}p}$ 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 $\mu Pa_{0\text{-}p}$ (Dalen and Knutsen 1986; Engås et al. 1996; Engås et al. 1993; Løkkeborg 1991; Løkkeborg and Soldal 1993; Turnpenny et al. 1994). Increased swimming activity in response to airgun exposure, as well as reduced foraging activity, is supported by data collected by Lokkeborg et al. (Løkkeborg et al. 2012). Bass did not appear to vacate during a shallow-water seismic survey with received sound levels of 163–191 dB re 1 $\mu Pa_{0\text{-}p}$ (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). La Bella et al. (1996b) found no differences in trawl catch data before and after seismic operations and echosurveys of fish occurrence did not reveal differences in pelagic biomass. However, fish kept in cages did show behavioral responses to approaching airguns.

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}. Guerra et al. (2004) suggested that giant squid mortalities were associated with seismic surveys based upon coincidence of carcasses with the surveys in time and space, as well as pathological information from the carcasses.

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.

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

Although leatherback sea turtles detect low frequency sound, the potential effects on sea turtle biology remain largely unknown (Samuel et al. 2005). 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 $_{rms}$ (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). However, loggerheads resting at the ocean surface were observed to frequently startle and dive as active seismic source approached them (DeRuiter and Larbi Doukara 2012). Responses decreased with increasing distance of closest approach by the seismic array (DeRuiter and Larbi Doukara 2012). The authors developed a response curve based upon observed responses and predicted received exposure level. 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 *Pelican* 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.

Changes in swimming patterns and potential temporary displacement is not expected to result in biologically meaningful separation from habitat. Loggerhead and leatherback sea turtles forage in wide geographical areas and movement to a secondary location is not expected to result in a meaningful metabolic or other physiological consequence for these wide-ranging species. Exceptions to this could be associations with artificial habitats, such as artificial platforms for loggerhead sea turtles. However, due to safety concerns, the seismic array would not closely approach such structures and potentially displace sea turtles from them.

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.

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, vessel 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 Atlantic 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 USGS proposes to conduct a seismic survey that could incidentally harass sperm whales. 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 (Table 3). 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 Gulf of Mexico face area-specific threats identified in the *Environmental Baseline*, including pollution.

Despite these pressures, available trend information indicates most local populations of listed sperm whales are stable or increasing. 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 sperm whales by proposed seismic activities. We expect up to 13 sperm whales could be exposed 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 is expected to represent a small fraction of the populations (not in excess of 0.2%, with some individual re-exposure) and reactions should not limit the fitness of any single individual. Behavioral harassment caused by exposure to sound sources associated with the proposed seismic survey are expected to cause some individuals to cease foraging, traveling, or social activities temporarily and possibly move out of the immediate area. However, we expect that individuals will either resume foraging in a secondary location or reoccupy the habitat from which they were displaced within a period of days. 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 leatherback sea turtle and loggerhead sea turtles, which of whom are both 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, naval activities, directed research, and climate change) to generally continue for the foreseeable future.

From the *Effects Analysis*, we expect that leatherback and loggerhead 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 or intensity of ensonifications (individually or cummulatively) 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. We do not anticipate any indirect effects from the proposed actions to influence sea turtles.

Considering the status of populations and species, the threat regimes individuals face, and the expected biological significance of stressors associated with the proposed action, we do not expect any individual sea turtle to undergo a fitness consequence. Many of the threats (such as bycatch and harvest) faced by sea turtles likely to be exposed to the proposed actions are highly significant (including traumatic injury or mortality when they do occur). These threats, when coupled with the small impact that the proposed seismic survey would have a on a limited percentage of population, are not expected to result in a fitness consequence at the individual, population, or species level. Other threats (climate change, research activities, pollution, etc.) are incremental that have impacts which are neither obvious nor, at present, readily predictable. However, is is reasonable that they can or will impair the fitness of significant portions of sea turtle populations. Based upon our understanding of these threats, individually, and cumulatively, we do not expect that the incremental addition of minor behavioral changes and ephemeral stress responses will result in fitness consequences as a result of the proposed action. Thus, we do not expect any sea turtle species to be jeopardized by the proposed action.

Conclusion

After reviewing the current status of sperm whales, leatherback sea turtles, and loggerhead 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 USGS' proposed seismic surveys in the Gulf of Mexico and NMFS Permits Division's issuance of an IHA 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 Division for harassment that would occur incidental to the proposed action is not likely to jeopardize the continued existence of these species. 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 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 USGS and NMFS' Permits Division so that they become binding conditions for USGS 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 term and conditions to implement the measures, must be provided. Only incidental take resulting from the agency actions and any specified reasonable and prudent measures and terms and conditions identified in the incidental take statement are exempt from the taking prohibition of Section 9(a), pursuant to Section 7(o) of the ESA.

Section 7(b)(4)(C) of the ESA specifies that in order to provide an incidental take statement for an endangered or threatened species of marine mammal, the taking must be authorized under Section 101(a)(5) of the MMPA. One of the federal actions considered in this Opinion is NMFS Permits Division's proposed authorization of the incidental taking of 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 Gulf of Mexico off Louisiana might result in the incidental take of listed species. The proposed action is expected to take 13 sperm whales by exposing individuals to received seismic sound levels greater than 160 dB re 1 μPa in the form of 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 *Pelican's* airgun array.

Harassment of sperm whales exposed to seismic studies at levels less than 160 dB re 1 μPa, or of leatherback and loggerhead 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 are in operation, this may constitute take that is not covered in this Incidental Take Statement. The USGS and NMFS' 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 sperm whales or leatherback sea turtles and loggerhead sea turtles is restricted to the permitted action as proposed. If the actual incidental take meets or exceeds the predicted level, the NSF and NMFS' Permits Division must reinitiate consultation. All anticipated takes would be in the form of 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 sperm whales and sea turtles resulting from the proposed action. These measures are non-discretionary and must be binding conditions of the USGS undertaking the proposed seismic studies and the NMFS' authorization for the exemption in Section 7(o)(2) to apply. If the USGS 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: vessel-based visual 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 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 USGS as well as Permits Division 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 USGS and the NMFS shall ensure that

1. USGS 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. USGS immediately reports all sightings and locations of injured or dead endangered and threatened species to NMFS' Permits Division.
4. The USGS and the NMFS' Permits 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 recommendation, 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 turtlesEffects of seismic noise on sea turtles. The USGS 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, NMFS' Permits 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 undertaken by the USGS on board the *R/V Pelican* in the Gulf of Mexico off Louisiana, 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, 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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