**NOAA's National Marine Fisheries Service**

**Endangered Species Act Section 7 Consultation Biological Opinion**

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| **Agencies:** | National Science Foundation-Division of Ocean Sciences,<br>United States Geological Survey, and<br>NOAA's National Marine Fisheries Service,<br>Office of Protected Resources,<br>Permits and Conservation Division |
| **Activities Considered:** | Seismic survey by the United States Geological Survey and Lamont-Doherty Earth Observatory along the U.S. East Coast and Issuance of an Incidental Harassment Authorization pursuant to Section 101(a)(5)(D) of the Marine Mammal Protection Act |
| **Consultation Conducted by:** | NOAA's National Marine Fisheries Service,<br>Office of Protected Resources,<br>Endangered Species Act Interagency Cooperation Division |
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| **Date:** | AUG 21 2014 |

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# List of Acronyms

CFR-Code of Federal Regulations

CI-confidence interval

CV-coefficient of variation

dB-decibel

DDE-dichlorodiphenyldichloroethylene

DDT-dichlorodiphenyltrichloroethane

DPS-distinct population segment

EEZ - Exclusive Economic Zone

ESA-Endangered Species Act

Hz-hertz

IHA-incidental harassment authorization

IWC-International Whaling Commission

kHz-kilohertz

kg-kilogram

L-DEO-Lamont Doherty Earth Observatory

MMPA-Marine Mammal Protection Act

NMFS-National Marine Fisheries Service

NMSDD-Navy Marine Species Density Database

NOAA-National Oceanic and Atmospheric Administration

NSF-National Science Foundation

OBIS-SEAMAP Ocean Biogeographic Information System Spatial Ecological Analysis of Megavertebrate Populations

Observer-protected species visual observer

PAM-passive acoustic monitoring

PBDE-polybrominated diphenyl ethers

PCB-polychlorinated biphenyl

PDE-phosphodiesterase

PFC-perflourinated chemicals

PFCA-perfluorinated carboxylic acids

PFOA-perfluorooctanoic acid

PFOS-perfluorooctanesulfonic acid

PIT-passive integrated transponder

PSI-pounds per square inch

PTS-permanent threshold shift

RMS-root mean squared

SBP-sub-bottom profiler

SCDNR-South Carolina Department of Natural Resources

SE-standard error

SEL-sound exposure level

TED-turtle excluder device

TEWG-Turtle Expert Working Group

TTS-temporary threshold shift

U.S. -United States

USC-United States Code

USGS-United States Geological Survey

# 1 INTRODUCTION

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" an ESA-listed species or critical habitat designated for it, that agency is required to consult with National Oceanic Atmospheric Administration's (NOAA) National Marine Fisheries Service (NMFS) or the U.S. Fish and Wildlife Service, depending upon the ESA-listed resources that may be affected. For the activities described in this document, the Federal action agencies are the United States Geological Survey (USGS), National Science Foundation (NSF), and NMFS's Permits and Conservation Division.

The NSF proposes to allow the use of its research vessel, *Marcus G. Langseth* (*Langseth*), which is operated by the Lamont-Doherty Earth Observatory (L-DEO), to conduct seismic surveys off the U.S. East Coast from August to September of 2014 and sometime between April and August 2015, in support of an NSF-funded collaborative research project led by the USGS. The NMFS's Permits and Conservation Division is also a Federal action agency as it is proposing to issue an incidental harassment authorization for non-lethal "takes" of marine mammals incidental to the planned seismic surveys, pursuant to Section 101 (a)(5)(D) of the Marine Mammal Protection Act (MMPA; 16 U.S.C. 1371 (a)(5)(D)). The consulting agency is the NMFS's Office of Protected Resources, ESA Interagency Cooperation Division.

This document represents NMFS's ESA 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 MMPA incidental harassment authorization application, draft public notice of proposed incidental harassment authorization, an environmental assessment prepared pursuant to the National Environmental Policy Act (NEPA), 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, biological opinions on similar activities, and other sources of information.

## 1.1 Consultation History

On November 4, 2013, the NMFS's ESA Interagency Cooperation Division received a request from the NMFS's Permits and Conservation Division for technical discussions on the proposed seismic survey. On Thursday, November 7, 2013, we met with the USGS, and its contractor responsible for USGS's environmental assessment, to discuss the proposed action. This initiated technical assistance on the action. Phone conversations among the entities continued on a regular basis here after.

On March 21, 2014, the NMFS's ESA Interagency Cooperation Division received a request for formal consultation from the USGS to incidentally harass marine mammal and sea turtle species during seismic surveys; information was not sufficient to initiate consultation with the USGS on this date. On the same date, the NMFS's Permits and Conservation Division received an

application from the L-DEO to incidentally harass marine mammal species during the proposed seismic survey.

On April 25, 2014, the NSF, USGS, and NMFS's ESA Interagency Cooperation Division discussed issues that were preventing initiation via teleconference.

On May 16, 2014, the USGS provided the NMFS's ESA Interagency Cooperation Division with an updated environmental assessment that, along with information that was previously provided, met ESA section 7 formal consultation initiation requirements. Remaining issues pertinent to assessing the effects of the action were resolved during other dates of the consultation.

On June 12, 2014, the NMFS's ESA Interagency Cooperation Division received a request for formal consultation from the NMFS's Permits and Conservation Division. Information was sufficient to initiate consultation with the Permits and Conservation Division on this date.

On June 23, 2014, the NMFS's Permits and Conservation Division sent the application for the proposed seismic surveys out to reviewers and published a notice in the *Federal Register* soliciting public comment on their intent to issue an incidental harassment authorization.

On August 15, 2014, the NMFS's ESA Interagency Cooperation Division received the final environmental assessment from the USGS.

# 2 DESCRIPTION OF THE PROPOSED ACTIONS

The NSF proposes to allow the use of its research vessel, *Marcus G. Langseth* (*Langseth*), which is operated by the L-DEO, to conduct a seismic survey off the U.S. East Coast during an approximate 21-day period in mid-August to September, 2014 and again during an unidentified three-week period from April to August 2015 in support of a USGS-funded and USGS-led research project. An array of 36 airguns will be deployed as an energy source. In addition, a multibeam echosounder and sub-bottom profiler (SBP) will continuously operate from the *Langseth*, except during transits to and from the survey site. An eight-kilometer-long hydrophone streamer will also be deployed as well as a towed magnetometer, 24 sonobuoys with suspended filament cable, and expendable bathythermographs (24 deployments). NMFS's Permits and Conservation Division proposes to issue an incidental harassment authorization for Level B harassment (behavioral disturbance) of marine mammals that would occur incidental to these studies, pursuant to Section 101(a)(5)(D) of the MMPA.

The purpose of the proposed activities is to 1) establish the outer limits of the U.S. continental shelf, also referred to as the Extended Continental Shelf , as defined by Article 76 of the Convention of the Law of the Sea and 2) study the sudden mass transport of sediments down the continental shelf as submarine landslides that pose potential tsunamigenic hazards to the Atlantic and Caribbean coastal communities. The proposed survey is part of a larger, multi-agency effort chaired by the U.S. Department of State and co-vice-chaired by Department of Interior and NOAA to determine the U.S. entitlement to sovereign rights in the area beyond 200 nautical miles according to established methods of measuring sediment thickness according to guidelines established by the Commission on the Limits of the Continental Shelf.

## 2.1 Schedule

The NSF proposes to allow the use of the *Langseth* by L-DEO in support of the USGS-led extended continental shelf activities for roughly 36-42 days of seismic operations, divided

between roughly equal operating periods in 2014 and 2015. Some minor deviation from the proposed dates is possible in 2014 and dates in 2015 have not yet been identified, although a definite period has been (April to August), depending on logistics and weather conditions. It is anticipated that portions of seismic lines may be reshot in the event of substandard data collection, but no additional line-kilometers of shooting would be added. This is because the reshooting effort would come at the expense of planned trackline being shot. During an approximate 21-day period in mid-August to September 2014 and again in April to August 2015, corresponding to an effective incidental harassment authorization, the *Langseth* would survey the action area. The *Langseth* would depart from Brooklyn, New York and return to Norfolk, Virginia. Therefore, NMFS's Permits and Conservation Division proposes to issue an authorization that is effective from August 15, 2014 to August 14, 2015. However, take would only be authorized during the period of August to September in 2014 and for a contiguous period of roughly 21 days between April and August 2015.

### 2.2 Source Vessel Specifications

The *Langseth* will tow the airgun array along predetermined lines (see Figure 1). The *Langseth's* design is that of a seismic research vessel, with a particularly quiet propulsion system to avoid interference with the seismic signals. The operating speed during seismic acquisition is typically 7.8-8.3 km/h (4.2-4.5 knots). When not towing seismic survey gear, the *Langseth* typically cruises at 20-24 km/h (11-12 knots).

The *Langseth* will also serve as the platform from which protected species visual observers (observers) would watch for animals.

### 2.3 Airgun Description

The airgun array will consist of 40 airguns (including four spare airguns), with a total operational volume of up to 6,600 $in^3$. However, only 36 of these airguns will be operational and total discharge volume will be limited to 6,600 $in^3$. The airgun configuration includes four identical linear arrays or "strings" (Figure 1). Each string will have ten airguns. Nine airguns in each of four strings would fire at any one time. The four airgun strings will be towed behind the vessel. The tow depth of the array will be 9 m. The airgun array will fire roughly every 22-23 seconds. During firing, a brief (approximately 0.1 s) pulse of sound will be emitted, followed by silence during the intervening listening. This signal attenuates as it moves away from the source, decreasing in amplitude, but also increasing in signal duration. Airguns will operate continually during the survey period except for unscheduled shut-downs.

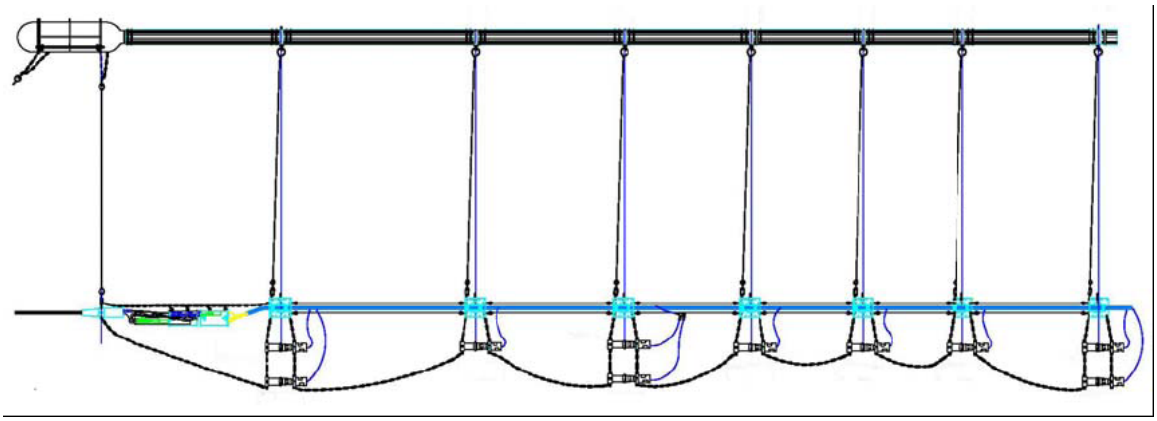

**Figure 1.** One linear airgun array or string with ten airguns.

**<u>36-airgun array specifications</u>**

| | |
|---|---|
| • Energy source | 36- 1,950 psi Bolt airguns of 40-360 $in^3$ each, in four strings of nine operating airguns per string |
| • Source output (downward)-36 airgun array | 0-pk is 259 dB re 1 μPa·m (84 bar·m); pk-pk is 265 dB re 1 μPa·m (177 bar·m) |
| • Air discharge volume | ~6,600 $in^3$ |
| • Dominant frequency components | 2–188 Hz |

Because the actual source originates from 36 airguns rather than a single point source, the highest sound levels measurable at any location in the water is less than the nominal source level. In addition, the effective source level for sound propagating in near-horizontal directions 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.

### 2.4 Multibeam Echosounder and Sub-bottom Profiler

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

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

The Knudsen Chirp 3260 SBP is normally operated to provide information about the sedimentary features and the bottom topography that is being mapped simultaneously by the MBES. The SBP is capable of reaching depths of 10,000 m. The beam is transmitted as a 27º cone, which is directed downward by a 3.5-kHz transducer in the hull of the *Langseth*. The nominal power output is 10 kW, but the actual maximum radiated power is 3 kW or 222 dB re 1

μPa·m. The ping duration is up to 64 ms, and the ping interval is 1 s. A common mode of operation is to broadcast five pings at 1-s intervals.

**<u>*Langseth* sub-bottom profiler specifications</u>**

| | |
|---|---|
| • Maximum/normal source output (downward) | 222 dB re 1 μPa·m |
| • Dominant frequency component | 3.5 kHz, up to 210 kHz |
| • Nominal beam width | 27° |
| • Ping duration | ≤64 ms |
| • Bandwidth | 1.0 kHz with pulse duration 4 ms<br>0.5 kHz with pulse duration 2 ms<br>0.25 kHz with pulse duration 1 ms |
| • Pulse duration | 1, 2, or 4 ms |

### 2.5 Proposed Exclusion Zones

The L-DEO will implement exclusion zones around the *Langseth* to minimize any potential adverse effects of airgun sound on MMPA and ESA-listed species. These zones are areas where seismic airguns would be powered down or shut-down to reduce exposure of marine mammals and sea turtles to sound levels expected to produce potential fitness consequences. These exclusion zones are based upon modeled sound levels at various distances from the *Langseth*, described below.

**Predicted Sound Levels vs. Distance and Depth**. The L-DEO has predicted received sound levels in deep water (free-field model), in relation to distance and direction from the 36-airgun array (Figure 2) as well as a 40-in$^3$ single 1900LLX airgun used during power-downs (Figure 3). Empirical data concerning 180 and 160 dB re 1 $\mu Pa_{rms}$ distances were acquired during the acoustic calibration study of the *Langseth*'s 36-airgun 6,600 in$^3$ array in the Gulf of Mexico (Diebold et al. 2010). However, the tow depth was different in the Gulf of Mexico calibration study (6 m tow depth) than in the proposed survey (9 m tow depth). Maximum radii were established at the maximum diving depth for listed species (2,000 m). As several species do not dive to this depth and, for those that do, we expect that individuals will rarely be found at this depth, the isopleth distance from the source array is likely to overestimate the exposure ESA-listed individuals are expected to experience.

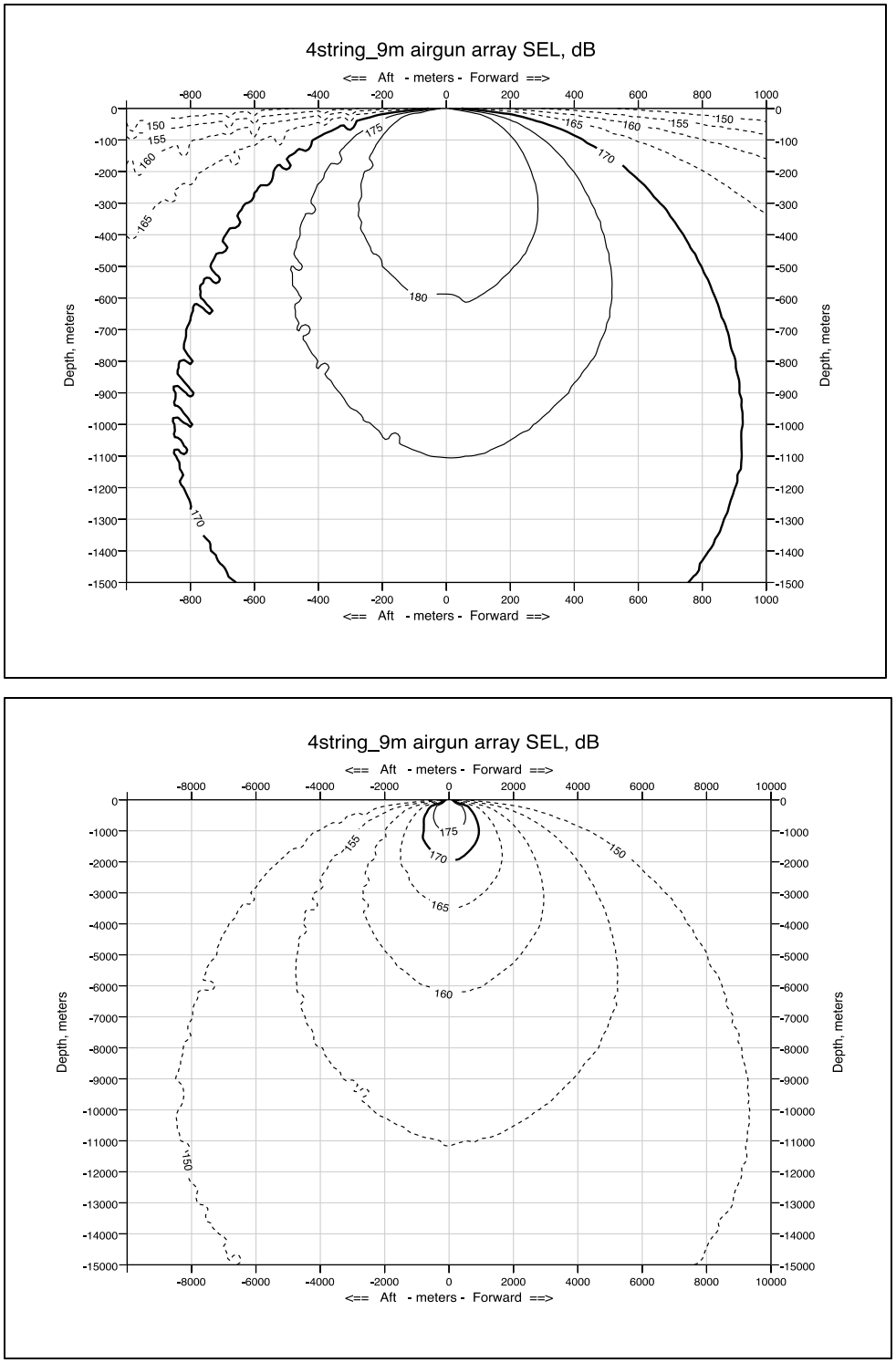


**Figure 2.** Modeled SEL contour distances for the 36-airgun array at nine meter tow depth in deep water.

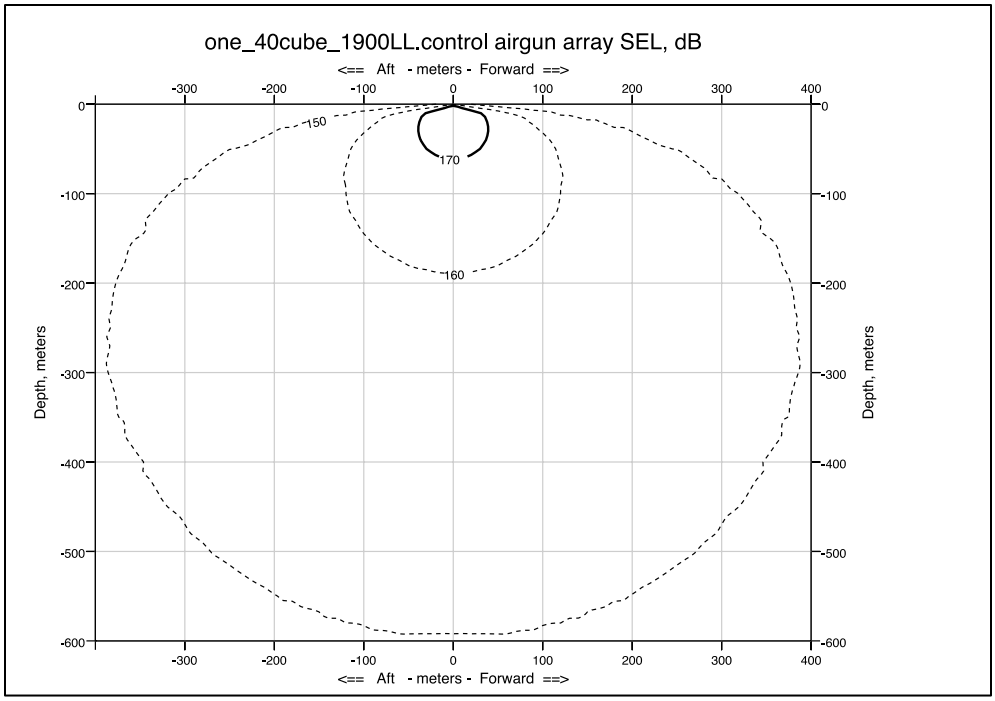


**Figure 3.** Modeled SEL contour distances for the 40 in$^3$ mitigation gun at nine meter tow depth in deep water.

Table 1 shows the distances at which three rms (root mean squared) sound levels are expected to be received from the 36-airgun array and a single airgun. The 180 dB re 1 μPa$_{rms}$ distance is the safety criteria as specified by NMFS (1995) as applicable to cetaceans under the MMPA. This will be used as the exclusion zone for marine mammals, as required by NMFS during most other recent L-DEO seismic projects (Cameron et al. 2013; Holst and Beland 2008; Holst and Smultea 2008b; Holst et al. 2005a; Holt 2008; L-DEO 2012; Smultea et al. 2004). The 180 dB isopleth would also be the exclusion zone boundary for sea turtles. The 166 dB isopleth represents our best understanding of the threshold at which sea turtles exhibit behavioral responses to seismic airguns. The 160 dB re 1 μPa$_{rms}$ distance is the distance at which MMPA take, by Level B harassment, is expected to occur.

**Table 1.** Predicted distances to which sound levels of 180, 166, and 160 dB re 1 μPa$_{rms}$ could be received from the 36-airgun arrays as well as the 40 in$^3$ airgun in water depths greater than 100 m.

| **Source, volume, and tow depth** | **Predicted RMS radii (m)** | | |
|---|---|---|---|
| | **180 dB** | **166 dB** | **160 dB** |
| 36-airgun array 6,600 in$^3$ @ 9 m | 927 | 3,740 | 5,780 |
| Single Bolt airgun, 40 in$^3$ @ 6 m | 100 | 185 | 388 |

### 2.6 Magnetometer, sonobuoys, and expendable bathythermographs

Several additional devices will be used during the proposed activities. A one-meter-long magnetometer will be deployed roughly 100 m behind the Langseth alongside the towed streamer. Twenty-four sonobuoys will be deployed from the side of the Langseth. These one-meter-long by 10 cm wide devices will float from the surface and drop a hydrophone 30 to 60 meters below the surface tethered by a filament cable. Deployments will last roughly eight hours before the sonobuoy sinks. Several dozen expendable bathythermographs will also be deployed from the side of the Langseth, deploying filaments to depths of several hundred to 1,000 m. Filament wire is relatively fragile and easily broken.

## 3 INCIDENTAL HARASSMENT AUTHORIZATION

The NMFS's Permits and Conservation Division is proposing to issue an incidental harassment authorization authorizing non-lethal "takes" by Level B harassment of marine mammals incidental to the planned seismic survey. The incidental harassment authorization will be valid from August 15, 2014 through August 14, 2015, and will authorize the incidental harassment of the following endangered species (among other species): blue whales (*Balaenoptera musculus*), fin whales (*Balaenoptera physalus*), sei whales (*Balaenoptera borealis*), humpback whales (*Megaptera novaeangliae*), North Atlantic right whale (*Eubalaena glacialis*), sperm whales (*Physeter macrocephalus*), and other non-listed marine mammals. Take would only be authorized from August to September 2014 and April to August 2015. The proposed incidental harassment authorization identifies the following requirements that L-DEO must comply with as part of its authorization.

A. Establish an exclusion zone[1] corresponding to the anticipated 180 dB re 1 $\mu Pa_{rms}$ isopleth for full (6,600 $in^3$) and single (40 $in^3$) airgun operations as well as a 160 dB re 1 $\mu Pa_{rms}$ buffer zone.

B. Use two, NMFS-approved, vessel-based observers to watch for and monitor marine mammal species near the seismic source vessel 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), optical range finders, and night vision devices. Observers shifts will last no longer than 4 hours at a time. Observers 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

1 The "exclusion zone" refers to a region around the seismic airgun source where mitigation would be undertaken to avoid or minimize the impacts of the airguns if marine mammals or sea turtles are observed within it.

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), Beaufort sea state and wind force, 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 exclusion zone using observers, for at least 30 min prior to starting the airgun (day or night). If observers find a marine mammal within the exclusion zone, USGS must delay the seismic survey until the marine mammal has left the area. If the observer sees a marine mammal that surfaces, then dives below the surface, the observer shall wait 30 minutes. If the observer sees no marine mammals during that time, they should assume that the animal has moved beyond the exclusion 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 exclusion zone, the airguns may not be started up. If one airgun is already running at a source level of at least 180 dB re 1 $\mu Pa_{rms}$, L-DEO may start subsequent guns without observing the entire exclusion zone for 30 min prior, provided no marine mammals are known to be near the safety radius. While it is considered unlikely, in the event a North Atlantic right whale (*Eubalaena glacialis*) is visually sighted, the airgun array will be shut-down regardless of the distance of the animal(s) to the sound source. The array will not resume firing until 30 min after the last documented whale visual sighting. Concentrations (greater than or equal to three individuals that do not appear to be traveling) of humpback, sei, fin, blue, and/or sperm whales will be avoided if possible (*i.e.*, exposing concentrations of animals to 160 dB), and the array will be powered-down if necessary.

E. Use the passive acoustic monitoring system (PAM) to detect marine mammals around the *Langseth* during all airgun operations and during most periods when airguns are not operating. One observer and/or bioacoustician will monitor the PAM at all times in shifts of 1-6 h. A bioacoustician shall design and set up the PAM system and be present to operate or oversee PAM, and be available if technical issues occur during the survey.

F. Record the following when an animal is detected by the PAM:

i. Contact the observer immediately (and initiate power or shut-down, if required);

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

G. Apply a “ramp-up” procedure when starting up at the beginning of seismic operations or any time after the entire array has been shut-down for more than 8 min, which means start the smallest gun first and add airguns in a sequence such that the source level of the array will increase in steps not exceeding approximately 6 dB per 5-min period. During ramp-up, the

observers will monitor the 180 dB re 1 $\mu Pa_{rms}$ exclusion zone, and if marine mammals are sighted, a course/speed alteration, power-down, or shut-down will occur as though the full array were operational.

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

I. Shut-down or power-down the airguns upon marine mammal detection within, approaching, or entering the exclusion zone. A power-down means shutting down one or more airguns and reducing the buffer and exclusion zones to the degree that the animal is outside of one or both. Following a power-down, if the marine mammal approaches the smaller designated exclusion zone, the airguns must be completely shut down. Airgun activity will not resume until the marine mammal has cleared the exclusion zone, which means it was visually observed to have left the exclusion zone, or has not been seen within the exclusion zone for 15 min (small odontocetes) or 30 min (mysticetes and large odontocetes). The *Langseth* may operate a small-volume airgun (*i.e.*, mitigation airgun) during turns and short maintenance periods (less than three hours) at approximately one shot per minute. During turns or brief transits between seismic tracklines, one mitigation airgun would continue to operate.

J. Marine seismic operations may continue into night and low-light hours if such segment(s) of the survey is initiated when the entire exclusion zone is 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 exclusion zone cannot be effectively monitored by the observer(s) on duty.

L. In the unanticipated event that the specified activity clearly causes any cases of marine mammal injury or mortality are judged to result from these activities (*e. g*., ship-strike, gear interaction, and/or entanglement), USGS will cease operating seismic airguns and report the incident to NMFS's Office of Protected Resources at 301-427-8401, and the NMFS Greater Atlantic Region Marine Mammal Stranding Network (866-755-6622) and/or by email to the NMFS Greater Atlantic Regional Stranding Coordinator (Mendy. Garron@noaa. gov), and the NMFS Southeast Region Marine Mammal Stranding Network (877-433-8299) and/or by e-mail to the Southeast Regional Stranding Coordinator (Blair. Mase@noaa. gov) and Southeast Regional Stranding Program Administrator (Erin. Fourgeres@noaa. gov) 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. If the lead observer judged that the injury or mortality is not a result of the authorized activities, operations may continue.

M. In the event that USGS discovers an injured or dead marine mammal, and the lead observer determines that the cause of the injury or death is unknown and the death is relatively recent (*i.e.*, in less than a moderate state of decomposition as described in the next paragraph), USGS will immediately report the incident to the Chief of the Permits and Conservation Division, Office of Protected Resources, NMFS, at 301-427-8401, and/or by email to Jolie.Harrison@noaa.gov and Howard.Goldstein@noaa.gov, and the NMFS Greater Atlantic Region Marine Mammal Stranding Network (866-755-6622) and/or by email to the NMFS Greater Atlantic Regional

Stranding Coordinator (Mendy.Garron@noaa.gov), and the NMFS Southeast Region Marine Mammal Stranding Network (877-433-8299) and/or by e-mail to the Southeast Regional Stranding Coordinator (Blair.Mase@noaa.gov) and Southeast Regional Stranding Program Administrator (Erin.Fourgeres@noaa.gov). Activities may continue while NMFS reviews the circumstances of the incident. NMFS will work with USGS to determine whether modifications in the activities are appropriate.

N. In the event that USGS discovers an injured or dead marine mammal, and the lead PSO determines that the injury or death is not associated with or related to the activities authorized in Condition 2 of this Authorization (e. g., previously wounded animal, carcass with moderate to advanced decomposition, or scavenger damage), USGS shall report the incident to the Chief of the Permits and Conservation Division, Office of Protected Resources, NMFS, at 301-427-8401, and/or by email to Jolie.Harrison@noaa.gov and Howard.Goldstein@noaa.gov, and the NMFS Greater Atlantic Marine Mammal Stranding Network (866-755-622), and/or by e-mail to the Greater Atlantic Regional Stranding Coordinator (Mendy.Garron@noaa.gov), and the NMFS Southeast Regional Stranding Network (877-433-8299), and/or by e-mail to the Southeast Stranding Coordinator (Blair.Mase@noaa.gov) and Southeast Regional Stranding Program Administrator (Erin.Fourgeres@noaa.gov), within 24 hours of the discovery. USGS shall provide photographs or video footage (if available) or other documentation of the stranded animal sighting to NMFS and the Marine Mammal Stranding Network. Activities may continue while NMFS reviews the circumstances of the incident.

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

In addition, the proposed incidental harassment authorization requires L-DEO to adhere to the following reporting requirements:

A. The Holder of this Authorization is required to submit a report on all activities and monitoring results to the Office of Protected Resources, NMFS, within 90 days after the completion of 2014 activities and again after the completion of 2015 activities. The report would describe the proposed operations that were conducted and sightings of marine mammals within the vicinity of the operations. The report would provide full documentation of methods, results, and interpretation pertaining to all monitoring. The 90-day report would summarize the dates and locations of seismic operations, and all marine mammal sightings (i.e., dates, times, locations, activities, associated seismic survey activities, and associated PAM detections). This report must also contain and summarize the following information:

1. Summaries of monitoring effort – total hours, total distances, and distribution of marine mammals through the study period accounting for Beaufort sea state and wind force, and other factors affecting visibility and detectability of marine mammals;

2. Analyses of the effects of various factors influencing detectability of marine mammals including Beaufort sea state and wind force, number of observers, and fog/glare;

3. Species composition, occurrence, and distribution of marine mammals sightings including date, water depth, numbers, age/size/gender, and group sizes; and

analyses of the effects of seismic operations;

4. Sighting rates of marine mammals during periods with and without airgun activities (and other variables that could affect detectability);
5. Initial sighting distances versus airgun activity state;
6. Closest point of approach versus airgun activity state;
7. Observed behaviors and types of movements versus airgun activity state;
8. Numbers of sightings/individuals seen versus airgun activity state; and
9. Distribution around the source vessel versus airgun activity state.
10. The report would also include estimates of the number and nature of exposures that could be used to further analyze and consider whether these were "takes" of marine mammals by harassment or in other ways.

# 4 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 ESA-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 ESA-listed resources that are likely to co-occur with these effects in space and time and the nature of that co-occurrence (these represent our *Exposure Analyses*). In this step of our analyses, we try to identify the number, age (or life stage), and gender of the individuals that are likely to be exposed to an action's effects and the populations or subpopulations those individuals represent. Once we identify which ESA-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 ESA-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 ESA-listed resources – are different for ESA-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 ESA-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 ESA-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 ESA-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 an individual is expected to experience reductions in fitness in response to an action's effects, those fitness reductions may 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 ESA-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 ESA-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 ESA-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 best scientific and commercial evidence available to us. This evidence consists of the environmental assessment submitted by the NSF, monitoring reports submitted by past and present seismic survey operators, 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 NMFS's Permits and Conservation Division when it initiates formal consultation, the general scientific literature, and our 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 also referred to an internal electronic library that represents a major repository on the biology of ESA-listed species under the NMFS's jurisdiction.

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 ESA-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 are 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 ESA-listed whales were multiplied by the area to be ensonified where effects were expected. Our primary concerns in this consultation revolve around exposure of listed individuals to anthropogenic sound sources, which can have a variety of effects that can have fitness consequences (Francis and Barber 2013; Nowacek and Tyack 2013) (Figure 4).

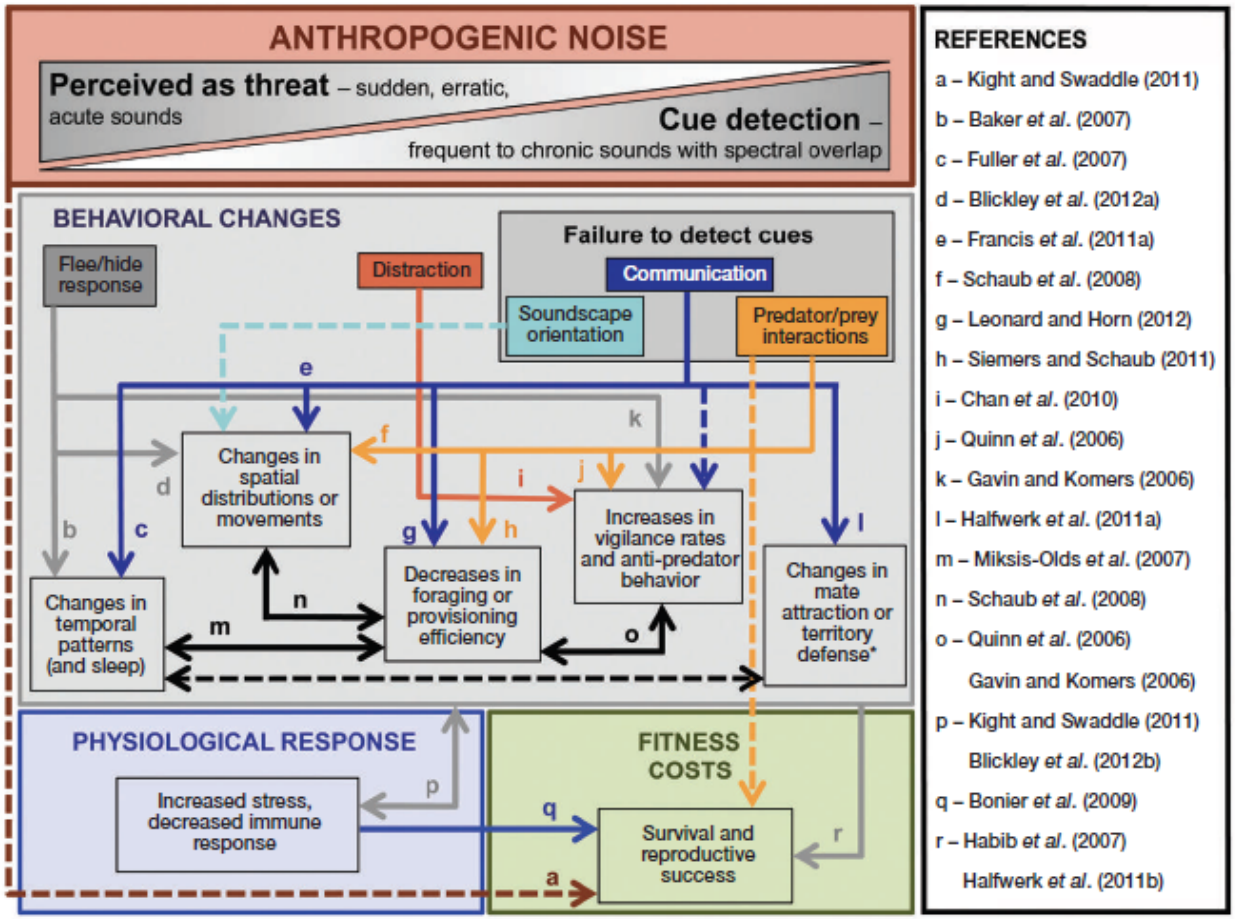


**Figure 4.** Conceptual framework of how anthropogenic noise impacts individuals and how those impacts can lead to fitness consequences. Figure taken from Francis et al. (2013). Original supporting literature (A. et al. 2007; Baker et al. 2007; Blickley et al. 2012; Blickley and Patricelli 2012; Bonier et al. 2009; Chan et al. 2010; D. et al. 2011; Gavin and Komers 2006; Habib et al. 2007; Halfwerk et al. 2011a; Halfwerk et al. 2011b; Kight and Swaddle 2011a; Leonard and Horn 2012; Miksis-Olds et al. 2007; Quinn et al. 2006; Schaub et al. 2008; Siemers and Schaub 2011).

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 (Southall et al. 2007a) better than high frequencies (Southall et al. 2007a), as the former is primarily the range in which they vocalize. Humpback whales frequently vocalize with mid-frequency sound (Southall et al. 2007a) 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; i.e., vocalization, as is assumed for baleen whales). 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).

# 5 ACTION AREA

The seismic survey is proposed to be conducted along a broad stretch of the U.S. East Coast (Figure 5), outside of state waters, and both within the U.S. Exclusive Economic Zone (EEZ) (10-20% of trackline) as well as outside of it (80-90%). The region in which the seismic survey will occur is between 29.2° and 39.2° N and 61.7° and 75.9° W. The region encompasses water depths from 1,450-5,400 m along roughly 1,707 km of trackline in 2014 and 1,682 km of trackline in 2015, respectively, including turns and other seismic operations. The applicant did not request an increase in trackline due to equipment failures, a need to reshoot some areas, or other logistical impacts. Responses to seismic sound sources by ESA-listed marine mammals occur within the 160 dB isopleths (modeled to be up to 5.780 km from the *Langseth*), increasing the area ensonified along the trackline to roughly 35,587 $km^2$ in 2014 and 2015, respectively. Responses to seismic sound sources by ESA-listed sea turtles occur within the 166 dB isopleths (modeled to be up to 3.740 km from the *Langseth*), increasing the area ensonified along the trackline to 12,768 $km^2$ in 2014 and 12,581 $km^2$ in 2015 (25,349 $km^2$ total). The transect lines are generally not close to one another, meaning that very few areas will be re-ensonified at high levels multiple times. We also assessed the transit to and from port for potential effects.

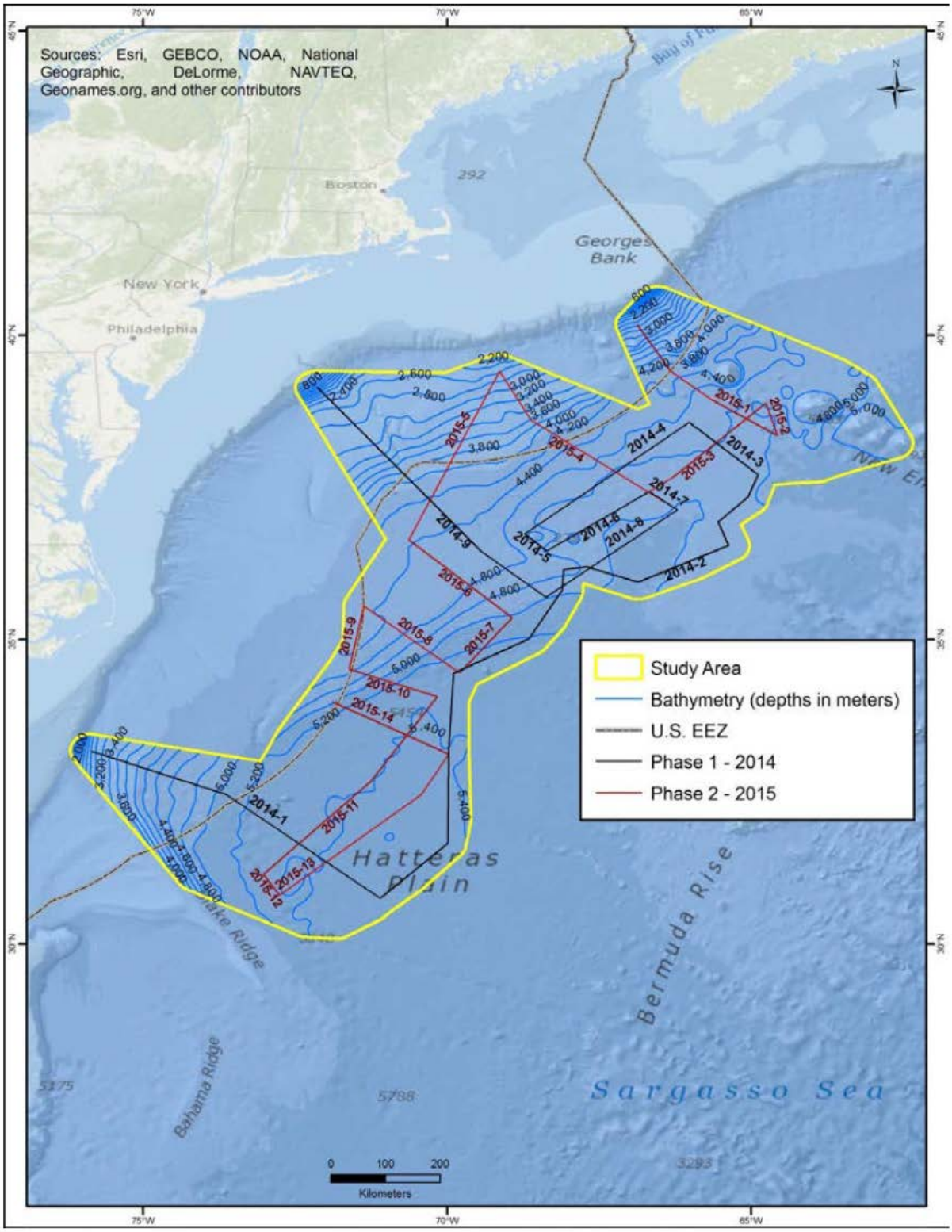


**Figure 5.** Proposed area for the marine seismic survey off the U.S. East Coast. Trackline for the seismic survey is identified in black and red lines for 2014 and 2015 activities, respectively. The

exclusion zone (area where mitigation would be undertaken if protected species are observed; not the U.S. EEZ) is not visible but occurs roughly one kilometer to either side of the trackline.

# 6 STATUS OF LISTED RESOURCES

The actions considered in this Opinion may affect species listed in Table 2, which are provided protection under the ESA.

**Table 2.** ESA-listed species in the action area that may experience adverse effects as a result of the proposed actions.

| *Common Name* | *Scientific Name* | *Status* |
|---|---|---|
| ***Cetaceans*** | | |
| Blue whale | *Balaenoptera musculus* | Endangered |
| Fin whale | *Balaenoptera physalus* | Endangered |
| Humpback whale | *Megaptera novaeangliae* | Endangered |
| North Atlantic right whale | *Eubalaena glacialis* | Endangered |
| Sei whale | *Balaenoptera borealis* | Endangered |
| Sperm whale | *Physeter macrocephalus* | Endangered |
| ***Marine Turtles*** | | |
| Green sea turtle | *Chelonia mydas* | Threatened |
| Hawksbill sea turtle | *Eretmochelys imbricata* | Endangered |
| Kemp's ridley sea turtle | *Lepidochelys kempii* | Endangered |
| Leatherback sea turtle | *Dermochelys coriacea* | Endangered |
| Loggerhead sea turtle – Northwest Atlantic distinct population segment | *Caretta caretta* | Endangered |

Although the area in which the seismic survey is proposed to occur is relatively close to shore, we do not believe that ESA-listed sturgeons are likely to be present in the action area. Both Atlantic and shortnose sturgeon occur in nearshore marine waters along the mid-Atlantic, but tagging studies have not found them to occur as far offshore as the proposed action area. We also do not expect Atlantic salmon to occur in the action area during the seismic survey. Thus, NMFS does not anticipate that the proposed seismic survey would incidentally take any ESA-listed sturgeons or Atlantic salmon. The action area also co-occurs with designated critical habitat of Northwestern Atlantic Distinct Population Segment (DPS) loggerhead sea turtles, specifically *Sargassum* habitat. The primary constituent elements of the critical habitat include: 1) convergence zones, surface-water downwelling areas, and other locations where there are concentrated components of the *Sargassum* community in water temperatures suitable for the optimal growth of *Sargassum* and inhabitance of loggerheads, 2) *Sargassum* in concentrations that support adequate prey abundance and cover, 3) available prey and other material associated with *Sargassum* habitat such as, but not limited to, plants and cyanobacteria and animals endemic to the *Sargassum* community such as hydroids and copepods, and 4) sufficient water depth (greater than 10 m) and proximity to available currents to ensure offshore transport, and foraging and cover requirements by *Sargassum* for post-hatchling loggerheads. We do not expect any stressors associated with the proposed actions to alter oceanographic or bathymetry features of the action area, impact the way in which *Sargassum* concentrates, or alter plant, cyanobacteria, or prey animals of loggerheads. Therefore, we do not expect any of the stressors of the proposed actions to impact the primary constituent elements of loggerhead critical habitat and, therefore, do not consider the critical habitat any further.

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.

### 6.1 Blue whale

**Subspecies.** Several blue whale subspecies have been characterized from morphological and geographical variability, but the validity of blue whale subspecies designations remains uncertain (McDonald et al. 2006). The largest, the Antarctic or true blue whale (*Balaenoptera musculus intermedia*), occurs in the highest Southern Hemisphere latitudes (Gilpatrick and Perryman. 2009). During austral summers, "true" blue whales occur close to Antarctic ice. A slightly smaller blue whale, *B. musculus musculus*, inhabits the Northern Hemisphere (Gilpatrick and Perryman. 2009). The pygmy blue whale (*B. musculus brevicauda*), may be geographically distinct from *B. m. musculus* (Kato et al. 1995). Pygmy blue whales occur north of the Antarctic Convergence (60°-80° E and 66°-70° S), while true blue whales are found south of the Convergence (58° S) in the austral summer (Kasamatsu et al. 1996; Kato et al. 1995). A fourth subspecies, *B. musculus indica*, may exist in the northern Indian Ocean (McDonald et al. 2006), although these whales are frequently referred to as *B. m. brevicauda* (Anderson et al. 2012). Inbreeding between *B. m. intermedia* and *B. m. brevicauda* does occur (Attard et al. 2012).

**Population structure.** Little is known about population and stock structure[2] of blue whales. Studies suggest a wide range of alternative population and stock scenarios based on movement, feeding, and acoustic data. Some suggest that as many as 10 global populations may exist, while other studies suggest that the species is composed of a single panmictic population (Gambell 1979; Gilpatrick and Perryman. 2009; Reeves et al. 1998). For management purposes, the International Whaling Commission (IWC) considers all Pacific blue whales to be a single stock, whereas under the MMPA, the NMFS recognizes four stocks of blue whales: western North Pacific Ocean, eastern North Pacific Ocean, Northern Indian Ocean, and Southern Hemisphere.

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

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

**North Atlantic.** Blue whales are found from the Arctic to at least mid-latitude waters, and typically inhabit the open ocean with occasional occurrences in the U.S. EEZ (Gagnon and Clark 1993; Wenzel et al. 1988; Yochem and Leatherwood 1985). Yochem and Leatherwood (1985) summarized records suggesting winter range extends south to Florida and the Gulf of Mexico. The U.S. Navy's Sound Surveillance System acoustic system has detected blue whales in much of the North Atlantic, including subtropical waters north of the West Indies and deep waters east of the U.S. EEZ (Clark 1995). Blue whales are rare in the shelf waters of the eastern U.S. In the western North Atlantic, blue whales are most frequently sighted from the Gulf of St. Lawrence and eastern Nova Scotia and in waters off Newfoundland, during the winter (Sears et al. 1987). In the eastern North Atlantic, blue whales have been observed off the Azores, although Reiner et al. (1993) did not consider them common in that area. Observations of feeding have recently occurred over Ireland's western continental slope (Wall et al. 2009). A single sighting was made in the study area 55 years ago, but several have been made in the region nearby to the northwest, particularly over the continental shelf break (Belford et al. 2014).

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

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

**Movement.** Satellite tagging indicates that, for blue whales tagged off Southern California, movement is more linear and faster (3.7 km/h) while traveling versus while foraging (1.7 km/h)(Bailey et al. 2009). Residency times in what are likely prey patches averages 21 days and constituted 29% of an individual's time overall, although foraging could apparently occur at any time of year for tagged individuals (Bailey et al. 2009). Broad scale movements also varied greatly, likely in response to oceanographic conditions influencing prey abundance and distribution (Bailey et al. 2009). Blue whales along Southern California were found to be traveling 85% of the time and milling 11% (Bacon et al. 2011). Blue whales are highly mobile, and their migratory patterns are not well known (Perry et al. 1999; Reeves et al. 2004). Blue whales migrate toward the warmer waters of the subtropics in fall to reduce energy costs, avoid ice entrapment, and reproduce (NMFS 1998a). In the eastern Central Atlantic, blue whales appear to migrate from areas along Greenland and Iceland to the Azores over and east of the Mid-Atlantic Ridge, apparently engaging in some random movement along the way (Anil et al. 2013).

**Feeding.** Data indicate that some summer feeding takes place at low latitudes in upwelling-modified waters, and that some whales remain year-round at either low or high latitudes (Clarke and Charif 1998b; Hucke-Gaete et al. 2004; Reilly and Thayer 1990; Yochem and Leatherwood 1985). Prey availability likely dictates blue whale distribution for most of the year (Burtenshaw et al. 2004; Clapham et al. 1999; Sears 2002 as cited in NMFS 2006a). The large size of blue whales requires higher energy requirements than smaller whales and potentially prohibits fasting Mate et al. (1999). Blue whales typically occur alone or in groups of up to five animals, although

larger foraging aggregations of up to 50 have been reported including aggregations mixed with other rorquals such as fin whales (Corkeron et al. 1999; Shirihai 2002). While feeding, blue whales show slowed and less obvious avoidance behavior then when not feeding (Sears et al. 1983 as cited in NMFS 2005b).

**Diving.** Blue whales spend greater than 94% of their time underwater (Lagerquist et al. 2000). Generally, blue whales dive 5-20 times at 12-20 sec intervals before a deep dive of 3-30 min (Croll et al. 1999; Leatherwood et al. 1976; Mackintosh 1965; Maser et al. 1981; Strong 1990; Yochem and Leatherwood 1985). Average foraging dives are 140 m deep and last for 7.8 min (Croll et al. 2001). Non-foraging dives are shallower and shorter, averaging 68 m and 4.9 min (Croll et al. 2001). However, dives of up to 300 m are known (Calambokidis et al. 2003). Nighttime dives are generally shallower (50 m). Blue whales near Sri Lanka averaged 18 sec between breaths during surfacing dives, but went an average of 640 sec during deep dives (de Vos et al. 2013).

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

**Vocalization and hearing.** Blue whales produce prolonged low-frequency vocalizations that include moans in the range from 12.5-400 Hz, with dominant frequencies from 16-25 Hz, and songs that span frequencies from 16-60 Hz that last up to 36 sec repeated every 1 to 2 min (see Cummings and Thompson 1971; Cummings and Thompson 1977; Edds-Walton 1997b; Edds 1982; McDonald et al. 1995a; Thompson and Friedl 1982). Berchok et al. (2006) examined vocalizations of St. Lawrence blue whales and found mean peak frequencies ranging from 17.0-78.7 Hz. Reported source levels are 180-188 dB re 1μPa, but may reach 195 dB re 1μPa (Aburto et al. 1997; Clark and Ellison 2004; Ketten 1998b; McDonald et al. 2001). Samaran et al. (2010) estimated Antarctic blue whale calls in the Indian Ocean at 179 ± 5 dB re 1 $\mu Pa_{rms}$ in the 17-30 Hz range and pygmy blue whale calls at 175± 1 dB re 1 $\mu Pa_{rms}$ in the 17-50 Hz range.

In temperate waters, intense bouts of long patterned sounds are very common from fall through spring, but these also occur to a lesser extent during the summer in high latitude feeding areas. Short sequences of rapid calls in the 30-90 Hz band are associated with animals in social groups. The seasonality and structure of long patterned sounds suggest that these sounds are male displays for attracting females, competing with other males, or both. The context for the 30-90 Hz calls suggests that they are communicative but not related to a reproductive function. Vocalizations attributed to blue whales have been recorded in presumed foraging areas, along migration routes, and during the presumed breeding season (Beamish and Mitchell 1971; Cummings et al. 1972; Cummings and Thompson 1971; Cummings and Thompson 1977; Cummings and Thompson 1994; Rivers 1997; Thompson et al. 1996).

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

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

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

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

Table 3 contains historic and current estimates of blue whales. Globally, blue whale abundance has been estimated at between 5,000-13,000 animals (COSEWIC 2002; Yochem and Leatherwood 1985), a fraction of the 200,000 or more that are estimated to have populated the oceans prior to whaling (Maser et al. 1981; U.S. Department of Commerce 1983). Consideration of the status of populations outside of the action area is important under the present analysis to determine the how the risk to the affected population(s) bears on the status of the species as a whole.

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

| Region | Population, stock, or study area | Pre-exploitation estimate | 95% CI | Current estimate | 95% CI | Source |
|---|---|---|---|---|---|---|
| Global | ~~ | 200,000 | ~~ | 11,200-13,000 | ~~ | (DOC 1983; Maser et al. 1981) |
| | ~~ | ~~ | ~~ | 5,000-12,000 | ~~ | (COSEWIC 2002) |
| North Atlantic | Basinwide | 1,100-1,500 | ~~ | 100-555 | ~~ | (Braham 1991; Gambell 1976) |
| | ~~ | | | 1,000-2,000 | | (Sigurjonsson 1995) |
| | NMFS-western North Atlantic stock | ~~ | ~~ | 440 | ~~ | (Waring et al. 2013) |
| | Central and northeast Atlantic | ~~ | ~~ | 855 | 351-1,589 | (Pike et al. 2009b) |

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

**North Atlantic.** Commercial hunting had a severe effect on blue whales, such that they remain rare in some formerly important habitats, notably in the northern and northeastern North Atlantic (Sigurjónsson and Gunnlaugsson 1990). Sigurjónsson and Gunnlaugsson (1990) estimated that at least 11,000 blue whales were harvested from all whaling areas from the late-nineteenth to mid-twentieth centuries.

Current trends are unknown, although an increasing annual trend of 4.9% was reported for 1969–1988 off western and southwestern Iceland (Sigurjónsson and Gunnlaugsson 1990). Sigurjónsson

and Gunnlaugsson (1990) concluded that the blue whale population had been increasing since the late 1950s. In the northeastern Atlantic, blue whales are most common west and south of Iceland and may be the largest concentration of blue whales in the North Atlantic (Pike et al. 2009b). In this area, the population may be recovering at a rate of 4-5% (Pike et al. 2009b). Punt (2010) estimated the rate of increase for blue whales in the central North Atlantic to be 9% annually (3.83 SE) between 1987 and 2001.

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

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

Ship strike remains a major concern for blue whales (Figure 6). Additional mortality from ship strikes probably goes unreported because the whales do not strand or, if they do, they do not always have obvious signs of trauma. Studies have shown that blue whales respond to approaching ships in a variety of ways, depending on the behavior of the animals at the time of approach, and speed and direction of the approaching vessel. While feeding, blue whales react less rapidly and with less obvious avoidance behavior than whales that are not feeding (Sears 1983).

Increasing noise in the ocean may impair blue whale behavior. Although available data do not presently support traumatic injury from sonar, the general trend in increasing ambient low-frequency noise in the deep oceans of the world, primarily from ship engines, could impair the ability of blue whales to communicate or navigate through these vast expanses (Aburto et al. 1997; Clark 2006). Blue whales off California altered call levels and rates in association with changes in local vessel traffic (McKenna 2011). Either due to ship strike, vessel noise, whale watching, or a combination of these factors, displacement from preferred habitat may be occurring off Sri Lanka (Ilangakoon 2012).

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

**Figure 6.** A near collision between a blue whale and a commercial cargo vessel in the Santa Barbara Channel Traffic Separation Scheme. Photo credit: NOAA Channel Islands National Marine Sanctuary, 2002 (Permit CINMS-2002-001).

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

### 6.2 Fin whale

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

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

**North Atlantic.** Fin whales are common off the Atlantic coast of the U.S. in waters immediately off the coast seaward to the continental shelf (about the 1,800 m contour). Fin whales occur during the summer from Baffin Bay to near Spitsbergen and the Barents Sea, south to Cape Hatteras in North Carolina and off the coasts of Portugal and Spain (Rice 1998a). In

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

Fin whales are also endemic to the Mediterranean Sea, where (at least in the western Mediterranean), individuals tend to aggregate during summer and disperse in winter over large spatial scales (Cotte et al. 2009), although this seasonal trend is reversed in the Bonifacio Strait (Arcangeli et al. 2013a). Mediterranean fin whales are genetically distinct from fin whales in the rest of the North Atlantic at the population level (Berube et al. 1999). However, some fin whales from the northeastern North Atlantic have been tracked into the Mediterranean during winter and overlap in time and space with the Mediterranean population may exist (Castellote et al. 2010). Individuals also tend to associate with colder, saltier water, where steep changes in temperature, and where higher northern krill densities would be expected (Cotte et al. 2009). A genetically distinct population resides year-round in the Ligurian Sea (IWC 2006). Fin whales seem to track areas of high productivity in the Mediterranean, particularly along coastal areas of France, northern Italy, and the southern and middle Adriatic (Druon et al. 2012). Several sightings have been within the study area, particularly in the northwestern sector, but also over the continental slope and abyssal plain (Belford et al. 2014). However, sightings are very common near the continental shelf break and over the continental shelf (Belford et al. 2014).

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

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

**Movement.** In the eastern Central Atlantic, fin whales appear to migrate from areas along Iceland to the Azores east of the Mid-Atlantic Ridge, apparently traveling directly without random movement patterns in between (Anil et al. 2013).

**Behavior.** Fin whales along Southern California were found to be traveling 87% of the time and milling 5% in groups that averaged 1.7 individuals (Bacon et al. 2011). Fin whales tend to avoid tropical and pack-ice waters, with the high-latitude limit of their range set by ice and the lower-latitude limit by warm water of approximately 15° C (Sergeant 1977). Fin whale concentrations generally form along frontal boundaries or mixing zones between coastal and oceanic waters, which corresponds roughly to the 200 m isobath (the continental shelf edge (Cotte et al. 2009; Nasu 1974)).

**Feeding.** Fin whales in the North Atlantic eat pelagic crustaceans (mainly krill and schooling fish such as capelin, herring, and sand lance (Borobia and Béland 1995; Christensen et al. 1992a; Hjort and Ruud 1929; Ingebrigtsen 1929; Jonsgård 1966; Mitchell 1974; Overholtz and Nicolas 1979; Sergeant 1977; Shirihai 2002; Watkins et al. 1984)). Fin whales frequently forage along cold eastern current boundaries (Perry et al. 1999). Feeding may occur in waters as shallow as 10 m when prey are at the surface, but most foraging is observed in high-productivity, upwelling, or thermal front marine waters (Gaskin 1972; Nature Conservancy Council 1979 as cited in ONR 2001; Panigada et al. 2008; Sergeant 1977). While foraging, fin whales in the Mediterranean Sea have been found to move through restricted territories in a convoluted manner (Lafortuna et al. 1999). Fin whales in the central Tyrrhenian Sea appear to ephemerally exploit the area for foraging during summer, particularly areas of high primary productivity (Arcangeli et al. 2013b).

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

**Vocalization and hearing.** Fin whales produce a variety of low-frequency sounds in the 10-200 Hz range (Edds 1988; Thompson et al. 1992a; Watkins 1981; Watkins et al. 1987b). Typical vocalizations are long, patterned pulses of short duration (0.5-2 s) in the 18-35 Hz range, but only males are known to produce these (Croll et al. 2002; Patterson and Hamilton 1964). Richardson et al. (1995b) reported the most common sound as a 1 sec vocalization of about 20 Hz, occurring in short series during spring, summer, and fall, and in repeated stereotyped patterns during winter. Au (2000b) reported moans of 14-118 Hz, with a dominant frequency of 20 Hz, tonal vocalizations of 34-150 Hz, and songs of 17-25 Hz (Cummings and Thompson 1994; Edds 1988; Watkins 1981). Source levels for fin whale vocalizations are 140-200 dB re 1μPa·m (Clark and Ellison. 2004; Erbe 2002b). The source depth of calling fin whales has been reported to be about 50 m (Watkins et al. 1987b). In temperate waters, intense bouts of long patterned sounds are very common from fall through spring, but also occur to a lesser extent during the summer in high latitude feeding areas (Clarke and Charif 1998a). Short sequences of

rapid pulses in the 20-70 Hz band are associated with animals in social groups (McDonald et al. 1995b). Each pulse lasts on the order of one second and contains twenty cycles (Tyack 1999).

Although their function is still debated, low-frequency fin whale vocalizations travel over long distances and may aid in long-distance communication (Edds-Walton 1997a; Payne and Webb 1971). During the breeding season, fin whales produce pulses in a regular repeating pattern, which have been proposed to be mating displays similar to those of humpbacks (Croll et al. 2002). These vocal bouts last for a day or longer (Tyack 1999). The seasonality and stereotype of the bouts of patterned sounds suggest that these sounds are male reproductive displays (Watkins et al. 1987a), while the individual counter-calling data of McDonald et al. (1995b) suggest that the more variable calls are contact calls. Some authors feel there are geographic differences in the frequency, duration and repetition of the pulses (Thompson et al. 1992b).

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

**Status and trends.** Fin whales were originally listed as endangered in 1970 (35 FR 18319), and this status continues since the inception of the ESA in 1973. Although fin whale population structure remains unclear, various abundance estimates are available (Table 4). Consideration of the status of populations outside of the action area is important under the present analysis to determine the how the risk to the affected population(s) bears on the status of the species as a whole. Historically, worldwide populations were severely depleted by commercial whaling, with more than 700,000 whales harvested in the twentieth century (Cherfas 1989a; Cherfas 1989b).

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

| Region | Population, stock, or study area | Pre-exploitation estimate | 95% CI | Recent estimate | 95% CI | Source |
|---|---|---|---|---|---|---|
| Global | ~~ | >464,000 | ~~ | 119,000 | ~~ | (Braham 1991) |
| North Atlantic | Basinwide | 30,000-50,000 | ~~ | ~~ | ~~ | (Sergeant 1977) |
| | ~~ | 360,000 | 249,000-481,000 | ~~ | ~~ | (Roman and Palumbi 2003) |
| | ~~ | | | >50,000 | | (Sigurjonsson 1995) |
| | Eastern North Atlantic | | | 25,000 | | (2009) circa 2001 |
| | Central and northeastern Atlantic | ~~ | ~~ | 30,000 | 23,000-39,000 | (IWC 2007) |
| | Western North Atlantic | ~~ | ~~ | 3,590-6,300 | ~~ | (Braham 1991) |
| | NMFS-western North Atlantic stock | ~~ | ~~ | 3,985 | CV=0.24 | (NMFS 2008; Waring et al. |

| Region | Population, stock, or study area | Pre-exploitation estimate | 95% CI | Recent estimate | 95% CI | Source |
|---|---|---|---|---|---|---|
| | | | | | | 2012) |
| | Northeastern U.S. Atlantic cont'l shelf | ~~ | ~~ | 2,200-5,000 | ~~ | (Hain et al. 1992; Waring et al. 2000) |
| | IWC-Newfoundland-Labrador stock | ~~ | ~~ | 13,253 | 0-50,139* | (IWC 1992) |
| | Bay of Biscay | | | 7,000-8,000 | | (Goujon et al. 1994) |
| | IWC-British Isles, Spain, and Portugal stock | 10,500 | 9,600-11,400 | 4,485 | 3,369-5,600 | (Braham 1991) |
| | ~~ | ~~ | ~~ | 17,355 | 10,400-28,900 | (Buckland et al. 1992) |
| | IWC-east Greenland to Faroe Islands | ~~ | ~~ | 22,000 | 16,000-30,000 | (IWC 2014) |
| | IWC-west Greenland stock | ~~ | ~~ | 4,500 | 1,900-10,000 | (IWC 2014) |
| | Mediterranean Sea | | | 3,583 | 2,130-6,027 | (Forcada 1996) |

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

**North Atlantic.** Over 48,000 fin whales were caught between 1860-1970 (Braham 1991). Although protected by the IWC, from 1988-1995 there have been 239 fin whales harvested from the North Atlantic. Recently, Iceland resumed whaling of fin whales despite the 1985 moratorium imposed by the International Whaling Commission (IWC). Vikingsson et al. (2009) concluded that actual numbers were likely higher due to negative bias in their analysis, and that the population(s) were increasing at 4% annually. The abundance of fin whales in the Baffin Bay-Davis Strait summer feeding area is believed to be increasing (Heide-Jorgensen et al. 2010).

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

**Anthropogenic threats.** Increased noise in the ocean stemming from shipping seems to alter the acoustic patterns of singing fin whales, possibly hampering reproductive parameters across wide

regions (Castellote et al. 2012).

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

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

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

### 6.3 Humpback whale

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

**North Atlantic.** Humpback whales range from the mid-Atlantic Bight and the Gulf of Maine across the southern coast of Greenland and Iceland to Norway in the Barents Sea. Whales migrate to the western coast of Africa (Waerebeek et al. 2013), the Cape Verde Islands, and the Caribbean Sea during the winter. Humpback whales aggregate in four summer feeding areas: Gulf of Maine and eastern Canada, west Greenland, Iceland, and Norway (Boye et al. 2010; Katona and Beard 1990; Smith et al. 1999). Four sightings have been within the study area, particularly in the northern half of the region (Belford et al. 2014). However, sightings are commonplace near the continental shelf break and over the continental shelf (Belford et al. 2014).

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

little genetic support for wide-scale interchange of individuals between ocean basins or across the equator.

**Distribution.** Humpback whales are a cosmopolitan species that occur in the Atlantic, Indian, Pacific, and Southern oceans. Humpback whales migrate seasonally between warmer, tropical or sub-tropical waters in winter months (where they breed and give birth to calves, although feeding occasionally occurs) and cooler, temperate or sub-Arctic waters in summer months (where they feed; (Gendron and Urban 1993). In both regions, humpback whales tend to occupy shallow, coastal waters. However, migrations are undertaken through deep, pelagic waters (Winn and Reichley 1985). Humpback whales wintering in the West Indies migrate relatively directly to the Gulf of Maine and areas around Iceland and Norway (Kennedy et al. 2013). Some individuals may not migrate, or species occurrence in foraging areas may extend beyond summer months (Van Opzeeland et al. 2013).

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

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

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

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

1987). In the Gulf of California, humpback whale dive durations averaged 3.5 min (Strong 1990). Because most humpback prey is likely found within 300 m of the surface, most humpback dives are probably relatively shallow. In Alaska, capelin are the primary prey of humpback and are found primarily between 92 and 120 m; depths to which humpbacks apparently dive for foraging (Witteveen et al. 2008).

**Feeding.** During the feeding season, humpback whales form small groups that occasionally aggregate on concentrations of food that may be stable for long-periods of times. Humpbacks use a wide variety of behaviors to feed on various small, schooling prey including krill and fish (Hain et al. 1982; Hain et al. 1995; Jurasz and Jurasz 1979; Weinrich et al. 1992; Witteveen et al. 2011). The principal fish prey in the western North Atlantic are sand lance, herring, and capelin (Kenney et al. 1985b). There is good evidence of some territoriality on feeding and calving areas (Clapham 1994; Clapham 1996; Tyack 1981). Humpback whales are generally believed to fast while migrating and on breeding grounds, but some individuals apparently feed while in low-latitude waters normally believed to be used exclusively for reproduction and calf-rearing (Danilewicz et al. 2009; Pinto De Sa Alves et al. 2009). Some individuals, such as juveniles, may not undertake migrations at all (Findlay and Best. 1995). Additional evidence, such as songs sung in northern latitudes during winter, provide additional support to plastic seasonal distribution (Smith and G.Pike 2009). Relatively high rates of resighting in foraging sites suggest whales return to the same areas year after year (Ashe et al. 2013; Kragh Boye et al. 2010). This trend appears to be maternally linked, with offspring returning to the same areas their mothers brought them to once calves are independent (Baker et al. 2013; Barendse et al. 2013). Humpback whales in foraging areas may forage largely or exclusively at night when prey are closer to the surface (Friedlaender et al. 2013).

**Vocalization and hearing.** Humpback whale vocalization is much better understood than is hearing. Different sounds are produced that correspond to different functions: feeding, breeding, and other social calls (Dunlop et al. 2008). Males sing complex sounds while in low-latitude breeding areas in a frequency range of 20 Hz to 4 kHz with estimated source levels from 144-174 dB (Au 2000b; Au et al. 2006; Frazer and Mercado 2000; Payne 1970; Richardson et al. 1995c; Winn et al. 1970). Both mature and immature males sing in breeding areas (Herman et al. 2013). Males also produce sounds associated with aggression, which are generally characterized as frequencies between 50 Hz to 10 kHz and having most energy below 3 kHz (Silber 1986; Tyack 1983). Such sounds can be heard up to 9 km away (Tyack and Whitehead 1983). Other social sounds from 50 Hz to 10 kHz (most energy below 3 kHz) are also produced in breeding areas (Richardson et al. 1995c; Tyack and Whitehead 1983). While in northern feeding areas, both sexes vocalize in grunts (25 Hz to 1.9 kHz), pulses (25-89 Hz), and songs (ranging from 30 Hz to 8 kHz but dominant frequencies of 120 Hz to 4 kHz) which can be very loud (175-192 dB re 1 μPa at 1 m; (Au 2000b; Erbe 2002a; Payne and Payne 1985; Richardson et al. 1995c; Thompson et al. 1986; Vu et al. 2012). However, humpbacks tend to be less vocal in northern feeding areas than in southern breeding areas, possibly due to foraging (Richardson et al. 1995c; Vu et al. 2012). During migration, social vocalizations are generated at 123 to 183 dB re 1 μPa at1 m with a median of 158 dB re 1 μPa at 1 m (Dunlop et al. 2013).

**Status and trends.** Humpback whales were originally listed as endangered in 1970 (35 FR 18319), and this status remains under the ESA. (Winn and Reichley 1985) argued that the global humpback whale population consisted of at least 150,000 whales in the early 1900s, mostly in the Southern Ocean. Consideration of the status of populations outside of the action area is

important under the present analysis to determine the risk to the affected population(s) bears on the status of the species as a whole. Table 5 provides estimates of historic and current abundance for ocean regions.

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

| Region | Population, stock, or study area | Pre-exploitation estimate | 95% CI | Recent estimate | 95% CI | Source |
|---|---|---|---|---|---|---|
| Global | ~~ | 1,000,000 | ~~ | ~~ | ~~ | (Roman and Palumbi 2003) |
| | | | | 10,000 | | (NMFS 1987) |
| North Atlantic | Basinwide | 240,000 | 156,000-401,000* | 11,570 | 10,005-13,135* | (Stevick et al. 2003a) |
| | ~~ | ~~ | ~~ | >5,500 | ~~ | (Sigurjonsson 1995) |
| | Basinwide-females | ~~ | ~~ | 2,804 | 1,776-4,463 | (Palsbøll et al. 1997) |
| | Basinwide-males | ~~ | ~~ | 4,894 | 3,374-7,123 | (Palsbøll et al. 1997) |
| | Western North Atlantic | ~~ | ~~ | 11,600 | 10,000-13,000 | (IWC 2014) |
| | Western North Atlantic from Davis Strait, Iceland, to the West Indies | >4,685* | ~~ | ~~ | ~~ | *circa 1865; (Mitchell and Reeves 1983) |
| | West Greenland | ~~ | ~~ | 2,154 | CV=0.36 | (Heide-Jorgensen et al. 2012) |
| | Iceland | ~~ | ~~ | 5,000 | ~~ | (Pike et al. 2009a) |
| | NMFS-Gulf of Maine stock | ~~ | ~~ | 847 | CV=0.55 | (Waring et al. 2012) |
| | NMFS-Gulf of Maine stock including portions of the Scotian Shelf | ~~ | ~~ | 902 | 177-1,627 | (Clapham et al. 2003) |
| | Barents and Norwegian Seas | ~~ | ~~ | 889 | 331-1,447* | (Øien 2001) *in* (Waring et al. 2004) |

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

**North Atlantic.** Historical estimates have ranged from 40,000-250,000 (Smith and G.Pike 2009). Smith and Reeves (2010) estimated that roughly 31,000 individuals were removed from the North Atlantic due to whaling since the 1600s. Estimates of animals on Caribbean

breeding grounds exceed 2,000 individuals (Balcomb III and Nichols 1982). Several researchers report an increasing trend in abundance for the North Atlantic population, which is supported by increased sightings within the Gulf of Maine feeding aggregation (Barlow 1997; Katona and Beard 1990; Smith et al. 1999; Waring et al. 2001). The rate of increase varies from 3.2-9.4%, with rates of increase slowing over the past two decades (Barlow 1997; Katona and Beard 1990; Stevick et al. 2003a). If the North Atlantic population has grown according to the estimated instantaneous rate of increase ($r$ = 0.0311), this would lead to an estimated 18,400 individual whales in 2008 (Stevick et al. 2003a). Punt (2010) estimated the rate of increase for humpback whales in the Gulf of Maine to be 6.3% annually (1.2 SE). Pike et al. (2009a) suggested that the eastern and northeastern waters off Iceland are areas of significant humpback utilization for feeding, estimating nearly 5,000 whales in 2001 and proposing an annual growth rate of 12% for the area. The authors suggest that humpback whales in the area had probably recovered from whaling. However, recent data suggest that the upward growth may have slowed or ceased around Iceland according to analysis of survey data there (Pike et al. 2010). The Gulf of Maine stock is estimated to be increasing at a rate of 3.1% annually (Waring et al. 2013). Humpback whales summering off West Greenland appear to be increasing at a rate of 9.4% annually (Heide-Jorgensen et al. 2012).

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

Parasites and biotoxins from red-tide blooms are other potential causes of mortality (Perry et al. 1999). The occurrence of the nematode *Crassicauda boopis* appears to increase the potential for kidney failure in humpback whales and may be preventing some populations from recovering (Lambertsen 1992). Studies of 14 humpback whales that stranded along Cape Cod between November 1987 and January 1988 indicate they apparently died from a toxin produced by dinoflagellates during this period. One-quarter of humpback whales of the Arabian Sea population show signs of tattoo skin disease, which may reduce the fitness of afflicted individuals (Baldwin et al. 2010).

**Anthropogenic threats.** Three human activities are known to represent major threats to humpback whales: whaling, commercial fishing, and shipping. Historically, whaling represented the greatest threat to every population of whales and was ultimately responsible for several species being listed as endangered.

Organochlorines, including PCB and DDT, have been identified in humpback whale blubber (Gauthier et al. 1997b). Higher PCB levels have been observed in western Atlantic waters versus Pacific waters along the United States and levels tend to increase with individual age (Elfes et al. 2010); eastern Atlantic individuals fall between these two in contaminant burden (Ryan et al. 2014) . Although humpback whales in the Gulf of Maine and off Southern California tend to have the highest PCB concentrations, overall levels are on par with other baleen whales, which are generally lower than odontocete cetaceans (Elfes et al. 2010). These contaminants are

transferred to young through the placenta, leaving newborns with contaminant loads equal to that of mothers before bioaccumulating additional contaminants during life and passing the additional burden to the next generation (Metcalfe et al. 2004). Contaminant levels are relatively high in humpback whales as compared to blue whales. Humpback whales feed higher on the food chain, where prey carry higher contaminant loads than the krill that blue whales feed on.

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

### 6.4 North Atlantic right whale

**Population.** All North Atlantic right whales compose a single population. Although not all individuals undergo the same migratory pattern, no subpopulation structuring has been identified.

**Distribution.** Right whales occur in sub-polar to temperate waters in all major ocean basins in the world, with a clear migratory pattern of high latitudes in summer and lower latitudes in winter (Cummings 1985; Perry et al. 1999; Rice 1998b). The historical range of North Atlantic right whales extended as far south as Florida and northwestern Africa, and as far north as Labrador, southern Greenland, Iceland, and Norway (Cummings 1985; Reeves et al. 1978; Rice 1998b). Recent sightings have been made through some of the broader historical range, including Iceland, Greenland, Norway, and the Azores (Hamilton et al. 2009; Hamilton et al. 2007; Jacobsen et al. 2004; Silva et al. 2012). Additional rare sightings have been made in the Gulf of Mexico (Moore and Clark 1963; Schmidly et al. 1972). Most sightings in the western North Atlantic are concentrated within five primary habitats or high-use areas: coastal waters of the southeastern U.S., Cape Cod and Massachusetts Bays, the Great South Channel, the Bay of Fundy, and the Scotian Shelf (Winn et al. 1986). In 1994, the first three of these areas were designated as critical habitat for the North Atlantic right whale.

North Atlantic right whales have been observed from the mid-Atlantic Bight northward through the Gulf of Maine year-round, but are primarily found along the northeast U.S. during summer and Florida during winter, with migratory routes in between. In New England, peak abundance of North Atlantic right whales in feeding areas occurs in Cape Cod Bay beginning in late winter. In early spring (late February to April), peak North Atlantic right whale abundance occurs in Jordan and Wilkinson Basins to the Great South Channel (Kenney et al. 1995; Nichols et al. 2008; Pace III and Merrick 2008). In late June and July, North Atlantic right whale distribution gradually shifts to the northern edge of Georges Bank. In late summer (August) and fall, much of the population is found in waters in the Bay of Fundy, the western Gulf of Maine and around Roseway Basin (Kenney et al. 2001; Kenney et al. 1995; Pace III and Merrick 2008; Winn et al. 1986). However, year-to-year variation in space and time are known and likely result from patchy prey distribution (Nichols et al. 2008). Variation in the abundance and development of suitable food patches appears to modify the general patterns of movement by reducing peak numbers, stay durations, and specific locales (Brown et al. 2001; Kenney 2001). In particular, large changes in the typical pattern of food abundance will dramatically change the general pattern of North Atlantic right whale habitat use (Kenney 2001). No sightings have been made in the study area where seismic surveys would be undertaken, although sightings just west over the continental shelf and continental shelf break regularly occur (Belford et al. 2014).

**Migration and movement.** North Atlantic right whales exhibit extensive migratory patterns, traveling along the eastern seaboard of the U.S. and Canada between calving grounds off Georgia and Florida to northern feeding areas off the northeast U.S. and Canada in March/April

and the reverse direction in November/December. The longest tracking of a North Atlantic right whale was a migration of 1,200 miles in 23 days the Bay of Fundy to Georgia (Mate and Baumgartner 2001). Migrations are typically within 30 nautical miles of the coastline and in waters less than 160 feet deep. Although this pattern is well-known, most of the population, particularly the males and non-pregnant females, is not found in the calving area and may not follow this pattern. It is unknown where the majority of the non-calving population spends the winter. Whales may remain in their foraging habitat during winter (Morano et al. 2012).

There have been a few recent sightings of North Atlantic right whales far offshore, including those from Dutch ships indicating some individuals occur between 40° and 50° N, in waters influenced by the North Atlantic Current (the broad, eastward-flowing extension of the Gulf Stream). Right whales have been sighted offshore (greater than 30 miles) during surveys flown off the coast of northeastern Florida and southeastern Georgia from 1996 to 2001. These include three sightings in 1996, one in 1997, 13 in 1998, six in 1999, 11 in 2000, and six in 2001 (within each year, some were repeat sightings). Mate et al. (1997) recorded radio-tagged animals making extensive movements from the Gulf of Maine into deeper waters off the continental shelf (Mate et al. 1997). The frequency with which North Atlantic right whales occur in offshore waters in the southeastern U.S. remains unclear. Occasionally, individuals are observed in distant locations, including the Gulf of Mexico, Bermuda, Azores, the Gulf of St. Lawrence, Newfoundland, Greenland, Iceland, and northern Norway (an area known as a historical North Atlantic right whale feeding area Silva et al. 2012; Smith et al. 2006). The Norwegian sighting (September 1992) represents one of only two sightings this century of a right whale in Norwegian waters, and the first since 1926. Together, these long-range matches indicate an extended range for at least some individuals and perhaps the existence of important habitat areas not presently well described.

**Reproduction, growth, and demography.** Data through the 1990s suggests that mean calving interval increased since 1992 from 3.67 years to more than five years, a significant trend that hampers North Atlantic right whale recovery (Best et al. 2001a; Kraus et al. 2007). This reproductive rate was approximately half that reported from studied populations of southern right whales (Best et al. 2001b). This has been attributed to several possible causes, including higher abortion or perinatal losses (Browning et al. 2009). An analysis of the age structure of North Atlantic right whales suggests that the population contains a smaller proportion of juvenile whales than expected, which may reflect lowered recruitment and/or high juvenile mortality (Best et al. 2001a; Hamilton et al. 1998). In addition, it is possible that the apparently low reproductive rate is due in part to unstable age structure or to reproductive senescence on the part of some females. However, knowledge on either factor is poor. Even though investment in calves is high for North Atlantic right whales, an incident of calf exchange (probably accidentally and soon after birth) and subsequent adoption through weaning has been found (Frasier et al. 2010). Although North Atlantic right whales historically separated from their calves within one year, a shift appears to have taken place around 2001 where mothers (particularly less experienced mothers) return to wintering grounds with their yearling at a much greater frequency (71% overall)(Hamilton and Cooper. 2010). The significance of this change is unknown.

Calves reach roughly three-quarters of their adult body size by the time they wean at 12 months, roughly doubling their original body size and gaining about 36 kg daily (Fortune et al. 2012).

**Habitat.** Available evidence from North Atlantic right whale foraging and habitat studies shows

that North Atlantic right whales focus foraging activities where physical oceanographic features such as water depth, current, and mixing fronts combine to concentrate copepods (Baumgartner et al. 2003; Davies et al. 2014; Mayo and Marx 1990; Murison and Gaskin 1989; Wishner et al. 1988a).

**Feeding.** North Atlantic right whales fast during the winter and feed during the summer, although some may opportunistically feed during migration. North Atlantic right whales use their baleen to sieve copepods from dense patches, found in highly variable and spatially unpredictable locations in the Bay of Fundy, Roseway Basin, Cape Cod Bay, the Great South Channel, and other areas off of northern U.S. and Canada (Pendleton et al. 2009). The primary prey of North Atlantic right whales is zooplankton, especially shrimp-like copepods such as *Calanus* (Beardsley et al. 1996; Kenney et al. 1985a). North Atlantic right whales feed largely by skimming these prey from the ocean surface (Mayo and Marx 1990; Pivorunas 1979), but may feed anywhere in the water column (Goodyear 1993; Watkins and Schevill 1976; Watkins and Schevill 1979; Winn et al. 1995). Feeding behavior has only been observed in northern areas and not on calving grounds or during migration (Kraus et al. 1993).

**Diving.** Although North Atlantic right whales are known to be primarily surface feeders, foraging dives frequently extend to the deepest layers of the water column (Baumgartner et al. 2003; Goodyear 1993; Mate et al. 1997). North Atlantic right whale feeding dives are characterized by a rapid descent from the surface to between 80 and 175 m, where dives level off and individuals remain for 5 to 14 min before rapidly ascending back to the surface (Baumgartner and Mate 2003). Dive depth has been shown to be strongly correlated with the depth of peak copepod abundance (Baumgartner and Mate 2003). Prolonged periods at the surface have been noted for mothers and calves (Baumgartner and Mate 2003). Shallow foraging dives in the Great South Channel average 2 min and 6 to 8 m (Winn et al. 1995). However, dives along the outer shelf average 7 min (CETAP 1982b). Although North Atlantic right whales are not champion divers, they can dive to over 300 m (Mate et al. 1992). Group size varies, but is generally less than one dozen and singletons and pairs are most frequently observed (Jefferson et al. 1993).

North Atlantic right whales produce a variety of calls from 159-192 dB re: 1 µPa while in surface active groups on breeding grounds (Tryonis et al. 2013).

**Vocalization and hearing.** Right whales vocalize to communicate over long distances and for social interaction, including communication apparently informing others of prey patch presence (Biedron et al. 2005; Tyson and Nowacek 2005). Vocalization patterns amongst all right whale species are generally similar, with six major call types: scream, gunshot, blow, up call, warble, and down call (McDonald and Moore 2002; Parks and Tyack 2005). A large majority of vocalizations occur in the 300-600 Hz range with up- and down sweeping modulations (Vanderlaan et al. 2003). Vocalizations below 200 Hz and above 900 Hz were rare (Vanderlaan et al. 2003). Calls tend to be clustered, with periods of silence between clusters (Vanderlaan et al. 2003). Gunshot bouts last 1.5 hours on average and up to seven hours (Parks et al. 2012a). Blows are associated with ventilation and are generally inaudible underwater (Parks and Clark 2007). Up calls are 100-400 Hz (Gillespie and Leaper 2001). Gunshots appear to be a largely or exclusively male vocalization (Parks et al. 2005b). Smaller groups vocalize more than larger groups and vocalization is more frequent at night (Matthews et al. 2001). Moans are usually produced within 10 m of the surface (Matthews et al. 2001). Up calls were detected year-round

in Massachusetts Bay except July and August and peaking in April (Mussoline et al. 2012). Individuals remaining in the Gulf of Maine through winter continue to call, showing a strong diel pattern of up call and gunshot vocalizations from November through January possibly associated with mating (Bort et al. 2011; Morano et al. 2012; Mussoline et al. 2012). Estimated source levels of gunshots in non-surface active groups are 201 dB re 1 µPa p-p (Hotchkin et al. 2011). While in surface active groups, females produce scream calls and males produce up calls and gunshot calls as threats to other males; calves (at least female calves) produce warble sounds similar top their mothers' screams (Parks et al. 2003; Parks and Tyack 2005). Source levels for these calls in surface active groups range from 137-162 dB rms re: 1 µPa-m, except for gunshots, which are 174-192 dB rms re: 1 µPa-m (Parks and Tyack 2005). Up calls may also be used to reunite mothers with calves (Parks and Clark 2007). Atlantic right whales shift calling frequencies, particularly of up calls, as well as increase call amplitude over both long and short-term periods due to exposure to vessel noise (Parks and Clark 2007; Parks et al. 2005a; Parks et al. 2007a; Parks et al. 2011a; Parks et al. 2010; Parks et al. 2012b; Parks et al. 2006). North Atlantic right whales respond to anthropogenic sound designed to alert whales to vessel presence by surfacing (Nowacek et al. 2003; Nowacek et al. 2004b).

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

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

North Atlantic right whales were formerly abundant, with an estimated 5,500 individuals present in the 16$^{th}$ century throughout the North Atlantic (Reeves 2001; Reeves et al. 2007). However, genetic evidence suggests a much larger historical population size of 112,000 individuals (95 % confidence interval 45,000–235,000)(Ruegg et al. 2013). A review of the photo-id recapture database in June 2006, indicated that only 313 individually recognized North Atlantic right whales were observed during 2001. Recent additions to the photo-ID catalog lead to a minimum population estimate of 444 individuals (Waring et al. 2013). This represents a nearly complete census, and the estimated minimum population size. However, no estimate of abundance with an associated coefficient of variation has been calculated for the population. Furthermore, 55% of fathers have not been genetically identified, suggesting the population may be significantly larger than presently thought (Frasier 2005). This also suggests the occurrence of right whales in as yet unidentified habitats (Frasier 2005). The population growth rate reported for the period 1986 to 1992 by Knowlton et al. (1994) was 2.5%, suggesting the stock was showing signs of slow recovery. However, work by Caswell et al. (1999) suggested that crude survival probability declined from about 0.99 in the early 1980's to about 0.94 in the late 1990s. Additional work conducted in 1999 showed that survival had indeed declined in the 1990s, particularly for adult females (Best et al. 2001a). Another workshop in September 2002 further confirmed the decline in this population (Clapham 2002). The best available estimate of population trajectory suggests

the population is increasing at a rate of 2.6% over the 1990-2009 timeframe (Waring et al. 2013).

**Natural threats.** Several researchers have suggested that the recovery of North Atlantic right whales has been impeded by competition with other whales for food (Rice 1974; Scarff 1986). Mitchell (1975) analyzed trophic interactions among baleen whales in the western North Atlantic and noted that the foraging grounds of North Atlantic right whales overlapped with the foraging grounds of sei whales. Both species feed preferentially on copepods. Mitchell (1975) argued that the North Atlantic right whale population had been depleted by several centuries of whaling before steam-driven boats allowed whalers to hunt sei whales; from this, he hypothesized that the decline of the right whale population made more food available to sei whales and helped their population to grow. He then suggested that competition with the sei whale population impedes or prevents the recovery of the right whale population. Shark predation has been repeatedly documented on right whales calves along the southeastern U.S., some of which may be fatal (Taylor et al. 2013).

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

**Anthropogenic threats.** Several human activities are known to threaten North Atlantic right whales: whaling, commercial fishing, shipping, and environmental contaminants. Historically, whaling represented the greatest threat to every population of right whales and was ultimately responsible for listing right whales as an endangered species. As its legacy, whaling reduced North Atlantic right whales to about 300 individuals in the western North Atlantic Ocean; the number of North Atlantic right whales in the eastern North Atlantic Ocean is probably much smaller, if present at all.

Concern also exists over climate change and its effect on the ability of North Atlantic right whales to recover (Greene et al. 2003b). Specifically, the variations in oceanography resulting from current shifts and water temperatures can significantly affect the occurrence of the North Atlantic right whale's primary food, copepod crustaceans. If climate changes such that current feeding areas cannot sustain North Atlantic right whales, the population may have to shift to reflect changes in prey distribution, pursue other prey types, or face prey shortage. Changes in calving intervals with sea surface temperature have already been documented for southern right whales (Leaper et al. 2006).

North Atlantic right whales, as with many marine mammals, are exposed to numerous toxins in their environment, many of which are introduced by humans. Levels of chromium in North Atlantic right whale tissues are sufficient to be mutagenic and cause cell death in lung, skin, or testicular cells and are a concern for North Atlantic right whale recovery (Chen et al. 2009; Wise et al. 2008). The organochlorines DDT, DDE, PCBs, dieldrin, chlordane, hexachlorobenzene, and heptachlor epoxide have been isolated from blubber samples and reported concentrations may underestimate actual levels (Woodley et al. 1991). Mean PCB levels in North Atlantic right whales are greater than any other baleen whale species thus far measured, although less than one-quarter of the levels measured in harbor porpoises (Gauthier et al. 1997a; Van Scheppingen et al. 1996). Organochlorines and pesticides, although variable in concentration by season, do not appear to currently threaten North Atlantic right whale health and recovery (Weisbrod et al.

2000). Flame retardants such as polybrominated diphenyl ethers (PBDEs) (known to be carcinogenic) have also been measured in North Atlantic right whales (Montie et al. 2010).

**Critical habitat.** Although no critical habitat occurs in the action area, critical habitat is designated for right whales in the North Atlantic. NMFS designated three areas in June 1994 as critical habitat for *Eubalaena glacialis* for feeding and calving (59 FR 28805). The critical habitats for feeding cover portions of the Great South Channel (east of Cape Cod), Massachusetts Bay and Cape Cod Bay, and Stellwagen Bank. Northern critical habitat was designated because of the concentration of right whales that feed in the area, apparently associated with complex oceanographic features that drive prey density and distribution. This area has come under considerable scrutiny within the past few years because of the concern over ship strikes in this area. Boston serves as a major port facility and vessels transiting to and from the port cross critical habitat where North Atlantic right whale mortality occurs. Shipping traffic has generally increased in the recent past and could be considered to degrade the habitat due to the additional mortality and injury risk now present in the area. Although voluntary regulations are in place, these are frequently ignored and mandatory regulations are under consideration. The southern critical habitats are along Georgia and northeastern Florida coasts (waters from the coast out 15 nautical miles between the latitudes of 31°15’ N and 30°15’ N and from the coast out five nautical miles between 30°15’ N and 28°00’ N). Southern critical habitat is designated to protected calving and breeding grounds for North Atlantic right whales, which generally calve and breed in shallow coastal waters. This critical habitat has generally fared better than northern critical habitat and significant degradation has not been clearly identified. Modeling efforts suggest water temperature and depth are driving factors for right whale occurrence along the coasts of Florida and Georgia during winter, some of which occur in designated critical habitat and some of which do not (Keller et al. 2012).

### 6.5 Sei whale

**Population designations.** The population structure of sei whales is unknown and populations herein assume (based upon migratory patterns) population structuring is discrete by ocean basin.

**North Atlantic.** In the western North Atlantic, a major portion of the sei whale population occurs in northern waters, potentially including the Scotian Shelf, along Labrador and Nova Scotia, south into the U.S. EEZ, including the Gulf of Maine and Georges Bank (Mitchell and Chapman 1977; Waring et al. 2004). These whales summer in northern areas before migrating south to waters along Florida, in the Gulf of Mexico, and the northern Caribbean Sea (Gambell 1985b; Mead 1977). Sei whales may range as far south as North Carolina. In the U.S. EEZ, the greatest abundance occurs during spring, with most sightings on the eastern edge of Georges Bank, in the Northeast Channel, and in Hydrographer Canyon (CETAP 1982c). In 1999, 2000, and 2001, the NMFS aerial surveys found sei whales concentrated along the northern edge of Georges Bank during spring (Waring et al. 2004). Surveys in 2001 found sei whales south of Nantucket along the continental shelf edge (Waring et al. 2004). During years of greater prey abundance (e. g., copepods), sei whales are found in more inshore waters, such as the Great South Channel (1987 and 1989), Stellwagen Bank (1986), and the Gulf of Maine (Payne et al. 1990a; Schilling et al. 1992). In the eastern Atlantic, sei whales occur in the Norwegian Sea, occasionally occurring as far north as Spitsbergen Island, and migrate south to Spain, Portugal, and northwest Africa (Gambell 1985b; Jonsgård and Darling 1977). Several sightings have been within the study area, particularly in the northwestern sector (Belford et al. 2014). However,

sightings are generally frequent near the continental shelf break (Belford et al. 2014).

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

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

**Feeding.** Sei whales are primarily planktivorous, feeding mainly on euphausiids and copepods, although they are also known to consume fish (Waring et al. 2006). In the Northern Hemisphere, sei whales consume small schooling fish such as anchovies, sardines, and mackerel when locally abundant (Konishi et al. 2009; Mizroch et al. 1984; Rice 1977).

**Vocalization and hearing.** Data on sei whale vocal behavior is limited, but includes records off the Antarctic Peninsula of broadband sounds in the 100-600 Hz range with 1.5 sec duration and tonal and upsweep calls in the 200-600 Hz range of 1-3 sec durations (McDonald et al. 2005). Source levels of 189 ±5.8 dB re 1 µPa at 1m have been established for sei whales in the northeastern Pacific (Weirathmueller et al. 2013). Differences may exist in vocalizations between ocean basins (Rankin and Barlow 2007b). The first variation consisted of sweeps from 100 to 44 Hz, over 1.0 sec. During visual and acoustic surveys conducted in the Hawaiian Islands in 2002, Rankin and Barlow (2007a) recorded 107 sei whale vocalizations, which they classified as two variations of low-frequency downswept calls. The second variation, which was more common (105 out of 107) consisted of low frequency calls which swept from 39 to 21 Hz over 1.3 sec. These vocalizations are different from sounds attributed to sei whales in the Atlantic and Southern Oceans but are similar to sounds that had previously been attributed to fin whales in Hawaiian waters. Vocalizations from the North Atlantic consisted of paired sequences (0.5-0.8 sec, separated by 0.4-1.0 sec) of 10-20 short (4 ms) FM sweeps between 1.5-3.5 kHz (Thomson and Richardson 1995).

**Status and trends.** The sei whale was originally listed as endangered in 1970 (35 FR 18319), and this status remained since the inception of the ESA in 1973. Consideration of the status of populations outside of the action area is important under the present analysis to determine the how risk the risk to the affected population(s) bears on the status of the species as a whole. Table 6 provides estimates of historic and current abundance for ocean regions.

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

| Region | Population, stock, or study area | Pre-exploitation estimate | 95% CI | Recent estimate | 95% CI | Source |
|---|---|---|---|---|---|---|
| **Global** | -- | >105,000 | -- | 25,000 | -- | (Braham 1991) |
| **North Atlantic** | Basinwide | -- | -- | >4000 | -- | (Braham 1991) |
| | ~~ | | | >13,500 | | (Sigurjonsson 1995) |
| | NMFS-Nova Scotia stock | -- | -- | 386 | -- | (NMFS 2008; Waring et al. 2012) |
| | Northeast Atlantic | -- | -- | 10,300 | 0.268 | (Cattanach et al. 1993) |

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

**North Atlantic.** No information on sei whale abundance exists prior to commercial whaling (Perry et al. 1999). Between 1966 and 1972, whalers from land stations on the east coast of Nova Scotia engaged in extensive hunts of sei whales on the Nova Scotia shelf, killing about 825 individuals (Mitchell and Chapman 1977). In 1974, the North Atlantic stock was estimated to number about 2,078 individuals, including 965 whales in the Labrador Sea group and 870 whales in the Nova Scotia group (Mitchell and Chapman 1977). In the northwest Atlantic, Mitchell and Chapman (1977) estimated the Nova Scotia stock to contain 1,393-2,248 whales; an aerial survey program conducted from 1978 to 1982 on the continental shelf and edge between Cape Hatteras, North Carolina, and Nova Scotia generated an estimate of 280 sei whales (CETAP 1982c). These two estimates are more than 30 years out of date and likely do not reflect the current true abundance; in addition, the Cetacean and Turtle Assessment Program estimate has a high degree of uncertainty and is considered statistically unreliable (Perry et al. 1999; Waring et al. 2004; Waring et al. 1999). The total number of sei whales in the U.S. Atlantic EEZ remains unknown (Waring et al. 2006). Rice (1977) estimated total annual mortality for adult females as 0.088 and adult males as 0.103.

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

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

Sei whales are known to accumulate DDT, DDE, and PCBs (Borrell 1993; Borrell and Aguilar

1987; Henry and Best 1983). Males carry larger burdens than females, as gestation and lactation transfer these toxins from mother to offspring.

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

### 6.6 Sperm whale

**Populations.** There is no clear understanding of the global population structure of sperm whales (Dufault et al. 1999). Recent ocean-wide genetic studies indicate low, but statistically significant, genetic diversity and no clear geographic structure, but strong differentiation between social groups (Lyrholm and Gyllensten 1998; Lyrholm et al. 1996; Lyrholm et al. 1999). Chemical analysis also suggest significant differences in diet for animals captured in different regions of the North Atlantic. However, vocal dialects indicate parent-offspring transmission that support differentiation in populations (Rendell et al. 2011). Therefore, population-level differences may be more extensive than are currently understood.

The IWC currently recognizes four sperm whale stocks: North Atlantic, North Pacific, northern Indian Ocean, and Southern Hemisphere (Dufault et al. 1999; Reeves and Whitehead 1997). The NMFS recognizes six stocks under the MMPA- three in the Atlantic/Gulf of Mexico and three in the Pacific (Alaska, California-Oregon-Washington, and Hawaii; (Perry et al. 1999; Waring et al. 2004)). Genetic studies indicate that movements of both sexes through expanses of ocean basins are common, and that males, but not females, often breed in different ocean basins than the ones in which they were born (Whitehead 2003). Sperm whale populations appear to be structured socially, at the level of the clan, rather than geographically (Whitehead 2003; Whitehead et al. 2008). Matrilinear groups in the eastern Pacific share nuclear DNA within broader clans, but North Atlantic matrilinear groups do not share this genetic heritage (Whitehead et al. 2012).

**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 whale 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. A long-term study of sperm whales along Dominica, West Indies supports 17 discreet groups habituating this area (Gero et al. 2013). 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). Hundreds of sightings have been made in the study area, ranging from the continental shelf break to abyssal waters (Belford et al. 2014). However, most sightings are in the northern and western portions of the area where seismic surveys will be conducted (Belford et al. 2014). The Mid-Atlantic Bight is considered a summer habitat for sperm whales (Palka 2006).

**Movement.** Mature males range between 70º N in the North Atlantic and 70º S in the Southern Ocean (Perry et al. 1999; Reeves and Whitehead 1997), whereas mature females and immature individuals of both sexes are seldom found higher than 50º N or S (Reeves and Whitehead 1997). In winter, sperm whales migrate closer to equatorial waters (Kasuya and Miyashita 1988; Waring et al. 1993) where adult males join them to breed. Males identified in the Azores have been resighted in Norwegian waters (Steiner et al. 2012). 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). 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. While deep water is their typical habitat, sperm whales are rarely found in waters less than 300 m in depth (Clarke 1956; Rice 1989a). Sperm whales have been observed near Long Island, New York, in water between 40-55 m deep (Scott and Sadove 1997). When they are found relatively close to shore, sperm whales are usually associated with sharp increases in topography where upwelling occurs and biological production is high, implying the presence of a good food supply (Clarke 1956). Such areas include oceanic islands and along the outer continental shelf.

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

**Reproduction.** Female sperm whales become sexually mature at an average of 9 years or 8.25-8.8 m (Kasuya 1991). Males reach a length of 10 to 12 m at sexual maturity and take 9-20 years to become sexually mature, but require another 10 years to become large enough to successfully breed (Kasuya 1991; Würsig et al. 2000). Mean age at physical maturity is 45 years for males

and 30 years for females (Waring et al. 2004). Adult females give birth after roughly 15 months of gestation and nurse their calves for 2-3 years (Waring et al. 2004). The calving interval is estimated to be every 4-6 years between the ages of 12 and 40 (Kasuya 1991; Whitehead et al. 2008). 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 one 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; 7-9 along Dominica) versus the Pacific (25-30 individuals)(Gero et al. 2013; Jaquet and Gendron 2009). Groups may be stable for long periods, such as for 80 days in the Gulf of California (Jaquet and Gendron 2009). Males start leaving these family groups at about six years of age, after which they live in “bachelor schools,” but this may occur more than a decade later (Pinela et al. 2009). The cohesion among males within a bachelor school declines with age. During their breeding prime and old age, male sperm whales are essentially solitary (Christal and Whitehead 1997).

**Diving.** Sperm whales are one of 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. 2000). 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 2006b). 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. Consideration of the status of populations outside of the action area is important under the present analysis to determine how the risk to the affected population(s) bears on the status of the species as a whole. Table 7 contains historic and current estimates of sperm whales. 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). Small changes in reproductive parameters, such as the loss of adult females, can significantly alter the population trajectory of sperm whale populations (Chiquet et al. 2013).

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

| Region | Population, stock, or study area | Pre-exploitation estimate | 95% CI | Recent estimate | 95% CI | Source |
|---|---|---|---|---|---|---|
| Global | ~~ | ~~ | ~~ | 900,000 | ~~ | (Würsig et al. 2000) |
| | ~~ | 1,110,000 | 672,000-1,512,000 | 360,000 | 105,984-614,016* | (Whitehead 2002) |
| North Atlantic | Basinwide-females | 224,800 | ~~ | 22,000 | ~~ | (Gosho et al. 1984; Würsig et al. 2000) |
| | Northeast Atlantic, Faroes, Iceland, and U.S. East coast | ~~ | ~~ | 13,190 | ~~ | (Whitehead 2002) |
| | NMFS-North Atlantic stock | >4,685* | ~~ | 4,804 | 1,226-8,382* | (Waring et al. 2012) |
| | Iceland | ~~ | ~~ | 1,234 | 823-1,645* | (Gunnlaugsson and Sigurjónsson 1990) |
| | Faroe Islands | ~~ | ~~ | 308 | 79-537* | (Gunnlaugsson and Sigurjónsson 1990) |
| | Norwegian Sea | ~~ | ~~ | 5,231 | 2,053-8,409* | (Christensen et al. 1992b) |
| | Northern Norway to Spitsbergen | 15,000 | ~~ | 2,548 | 1,200-3,896* | (Øien 1990) |

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

**North Atlantic.1**90,000 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). 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 2006b; Townsend 1935).

**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). 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 2004a).

Whale-watching vessels are known to influence sperm whale behavior (Richter et al. 2006).

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, hexachlorobenzene and hexachlorocyclohexane 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.

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

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

### 6.7 Green sea turtle

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

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

| Location | Most recent abundance | Reference |
|---|---|---|
| **Western Atlantic Ocean** | | |
| Tortuguero, Costa Rica | 17,402-37,290 AF | (Troëng and Rankin 2005) |
| Aves Island, Venezuela | 335-443 AF | (Vera 2007) |
| Galibi Reserve, Suriname | 1,803 AF | (Weijerman et al. 1998) |
| Isla Trindade, Brazil | 1,500-2,000 AF | (Moreira and Bjorndal 2006) |
| **Central Atlantic Ocean** | | |
| Ascension Island, UK | 3,500 AF | (Broderick et al. 2006) |
| **Eastern Atlantic Ocean** | | |
| Poilao Island, Guinea-Bissau | 7,000-29,000 AN | (Catry et al. 2009) |
| Bioko Island, Equatorial Guinea | 1,255-1,681 AN | (Tomas et al. 1999) |
| **Mediterranean Sea** | | |
| Turkey | 214-231 AF | (Broderick et al. 2002) |
| Cyprus | 121-127 AF | (Broderick et al. 2002) |
| Israel / Palestine | 1-3 AF | (Kuller 1999) |
| Syria | 100 AN | (Rees et al. 2005) |

**Distribution.** Green sea turtles have a circumglobal distribution, occurring throughout tropical, subtropical waters, and, to a lesser extent, temperate waters. Several sightings have been made in the area of the proposed seismic survey tracklines (generally in the northwestern component of the area), but few during the same season as the survey (Belford et al. 2014). Summer sightings are generally over the continental shelf break (Belford et al. 2014).

**Growth and reproduction.** Most green sea turtles exhibit particularly slow growth rates, which have been attributed to their largely plant-eating diet (Bjorndal 1982). Growth rates of juveniles vary substantially among populations, ranging from <1 cm/year (Green 1993) to >5 cm/year (McDonald Dutton and Dutton 1998), likely due to differences in diet quality, duration of foraging season (Chaloupka et al. 2004), and density of turtles in foraging areas (Balazs and Chaloupka 2004; Bjorndal et al. 2000; Seminoff et al. 2002b). Hart et al. (2013a) found growth rates of green sea turtles in the U.S. Virgin Islands to range from 0-9.5 cm annually (mean of 4.1, SD 2.4). The largest growth rates were in the 30-39 cm class. If individuals do not feed

sufficiently, growth is stunted and apparently does not compensate even when greater-than-needed resources are available (Roark et al. 2009). In general, there is a tendency for green sea turtles to exhibit monotonic growth (declining growth rate with size) in the Atlantic and non-monotonic growth (growth spurt in mid-size classes) in the Pacific, although this is not always the case (Balazs and Chaloupka 2004; Chaloupka and Musick 1997; Seminoff et al. 2002b). It is estimated that green sea turtles reach a maximum size just under 100 cm in carapace length (Tanaka 2009). A female-bias has been identified from studies of green sea turtles (Wibbels 2003).

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

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

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

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

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

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

**Diving.** Based on the behavior of post-hatchlings and juvenile green turtles raised in captivity, we presume that those in pelagic habitats live and feed at or near the ocean surface, and that their dives do not normally exceed 7 m in depth (Hazel et al. 2009; NMFS and USFWS 1998). Recent data from Australia indicate green sea turtles rarely dive deep, staying in upper 8 m of the water column (Hazel et al. 2009). Here, daytime dives were shorter and shallower than were nighttime dives. Also, time spent resting and dive duration increased significantly with decreases in seasonal water temperatures. The maximum recorded dive depth for an adult green turtle was just over 106 m (Berkson 1967), while subadults routinely dive to 20 m for 9-23 min, with a maximum recorded dive duration of over 1 h (Brill et al. 1995; I-Jiunn 2009). Green sea turtles along Taiwan may rest during long, shallow dives (I-Jiunn 2009). Dives by females may be shorter in the period leading up to nesting (I-Jiunn 2009).

**Vocalization and hearing.** Sea turtles are low-frequency hearing specialists, typically hearing frequencies from 30 to 2,000 Hz, with a range of maximum sensitivity between 100 and 800 Hz (Bartol et al. 1999; Lenhardt 2002; Lenhardt 1994a; Moein Bartol and Ketten 2006; Ridgway et al. 1969). Piniak et al. (2012) found green sea turtle juveniles capable of hearing underwater sounds at frequencies of 50-1,600 Hz (maximum sensitivity at 200-400 Hz). Hearing below 80 Hz is less sensitive but still possible (Lenhardt 1994a). Based upon auditory brainstem responses green sea turtles have been measured to hear in the 50-1600 Hz range (Dow et al. 2008), with greatest response at 300 Hz (Yudhana et al. 2010); a value verified by Moein Bartol and Ketten (2006). Other studies have found greatest sensitivities are 200-400 Hz for the green turtle with a range of 100-500 Hz (Moein Bartol and Ketten 2006; Ridgway et al. 1969) and around 250 Hz or below for juveniles (Bartol et al. 1999). However, Dow et al. (2008) found best sensitivity between 50 and 400 Hz.

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

**Status and trends.** Federal listing of the green sea turtle occurred on July 28, 1978, with all populations listed as threatened except for the Florida and Pacific coast of Mexico breeding populations, which are endangered (43 FR 32800).

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

Based on the mean annual reproductive effort, 108,761-150,521 females nest each year among 46 worldwide sites. Overall, of the 26 sites for which data enable an assessment of current trends, 12 nesting populations are increasing, 10 are stable, and four are decreasing. Long-term continuous datasets of 20 years are available for 11 sites, all of which are either increasing or

stable. Despite the apparent global increase in numbers, the positive overall trend should be viewed cautiously because trend data are available for just over half of all sites examined and very few data sets span a full green sea turtle generation (Seminoff 2004a).

Long-term capture rates have increased exponentially for green sea turtles in the Laguna Madre of Texas from 1991-2010, although average size seems to be declining (Metz and Landry Jr. 2013). These trends may be due to increasing nest output from Mexican and Florida beaches, with juveniles recruiting into the neritic Texas coast (Metz and Landry Jr. 2013). Similarly, average turtle length has declined over the course of a long-term study along cape Canaveral, Florida, as has recapture rate, likely for the same reasons (Redfoot and Ehrhart 2013).

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

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

The vast majority of green sea turtle nesting within the southeastern U.S. occurs in Florida (Johnson and Ehrhart 1994; Meylan et al. 1995). Green sea turtle nesting in Florida has been increasing since 1989 (Florida Fish and Wildlife Conservation Commission, Florida Marine Research Institute Index Nesting Beach Survey Database). Since establishment of index beaches in 1989, the pattern of green turtle nesting shows biennial peaks in abundance with a generally positive trend during the ten years of regular monitoring. This is perhaps due to increased protective legislation throughout the Caribbean (Meylan et al. 1995). A total statewide average (all beaches, including index beaches) of 5,039 green turtle nests were laid annually in Florida between 2001 and 2006, with a low of 581 in 2001 and a high of 9,644 in 2005 (NMFS and USFWS 2007a). Data from index nesting beaches substantiate the dramatic increase in nesting. In 2007, there were 9,455 green turtle nests found just on index nesting beaches, the highest since index beach monitoring began in 1989. The number fell back to 6,385 in 2008, further dropping under 3,000 in 2009, but that consecutive drop was a temporary deviation from the normal biennial nesting cycle for green turtles, as 2010 saw an increase back to 8,426 nests on the index nesting beaches (FWC Index Nesting Beach Survey Database). Nesting in 2010 and

2011 increased again, decreased in 2012, and greatly increased in 2013 to more than double the previous high in 2011 (roughly 10,000)(FWC Index Nesting Beach Survey Database). From 1989-2013, green sea turtle nest counts across Florida have increased approximately ten-fold from a low of 267 in the early 1990s to a high of 25,553 in 2013 (FWC Index Nesting Beach Survey Database).

Occasional nesting has been documented along the Gulf coast of Florida (Meylan et al. 1995). More recently, green turtle nesting occurred on Bald Head Island, North Carolina; just east of the mouth of the Cape Fear River; on Onslow Island; and on Cape Hatteras National Seashore. In 2010, a total of 18 nests were found in North Carolina, six nests in South Carolina, and six nests in Georgia (nesting databases maintained on www. seaturtle. org). Increased nesting has also been observed along the Atlantic coast of Florida, on beaches where only loggerhead nesting was observed in the past (Pritchard 1997). Recent modeling by Chaloupka et al. (2008a) using data sets of 25 years or more has resulted in an estimate of the Florida nesting stock at the Archie Carr National Wildlife Refuge growing at an annual rate of 13.9%, and the Tortuguero, Costa Rica, population growing at 4.9%.

In Florida, index beaches were established to standardize data collection methods and effort on key nesting beaches. Since establishment of the index beaches in 1989, the pattern of green sea turtle nesting has generally shown biennial peaks in abundance with a positive trend during the 10 years of regular monitoring. According to data collected from Florida's index nesting beach survey from 1989-2012, green sea turtle nest counts across Florida have increased approximately ten-fold from a low of 267 in the early 1990s to a high of 25,553 in 2013. Two consecutive years of nesting declines in 2008 and 2009 caused some concern, but this was followed by increases in both 2010 and 2011, a decrease in 2012, and another increase in 2013. Modeling by Chaloupka et al. (2008b) using data sets of 25 years or more has resulted in an estimate of the Florida nesting stock at the Archie Carr National Wildlife Refuge growing at an annual rate of 13.9%.

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

**Natural threats.** Herons, gulls, dogfish, and sharks prey upon hatchlings. Adults face predation primarily by sharks and to a lesser extent by killer whales. Predators (primarily of eggs and hatchlings) also include dogs, pigs, rats, crabs, sea birds, reef fishes, and groupers (Bell et al. 1994; Witzell 1981).

For unknown reasons, the frequency of a disease called fibropapillomatosis is much higher in green sea turtles than in other species and threatens a large number of existing subpopulations. Extremely high incidence has been reported in Hawaii, where affliction rates peaked at 47-69% in some foraging areas (Murakawa et al. 2000). A to-date unidentified virus may aid in the development of fibropapillomatosis (Work et al. 2009). Green sea turtles with an abundance of

barnacles have been found to have a much greater probability of having health issues (Flint et al. 2009). The fungal pathogens *Fusarium falciforme* and *F. keratoplasticum* can kill in excess of 90% of sea turtle embryos they infect and may constitute a major threat to nesting productivity under some conditions (Sarmiento-Ramırez et al. 2014).

All sea turtles except leatherbacks can undergo "cold stunning" if water temperatures drop below a threshold level, which can be lethal.

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

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

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

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). 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 2010; Vargo et al. 1986a; Vargo et al. 1986b; 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 2010). 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). 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).

**Critical habitat.** On September 2, 1998, critical habitat for green sea turtles was designated in coastal waters surrounding Culebra Island, Puerto Rico (63 FR 46693). Aspects of these areas that are important for green sea turtle survival and recovery include important natal development habitat, refuge from predation, shelter between foraging periods, and food for green sea turtle prey. The proposed action does not co-occur with this critical habitat.

## 6.8 Hawksbill sea turtle

**Populations.** Populations are distinguished generally by ocean basin and more specifically by nesting location. Our understanding of population structure is relatively poor. For example, genetic analysis of hawksbill sea turtles foraging off the Cape Verde Islands identified three closely-related haplotypes in a large majority of individuals sampled that did not match those of any known nesting population in the western Atlantic, where the vast majority of nesting has been documented (McClellan et al. 2010; Monzon-Arguello et al. 2010). Hawksbills in the Caribbean seem to have dispersed into separate populations (rookeries) after a bottleneck roughly 100,000-300,000 years ago based upon genetic data (Leroux et al. 2012). Nesting in the northwestern Hawaiian Islands has been rarely found (partly stemming from poor observer effort), but is believed to have been greater historically (Van Houtan et al. 2012).

**Distribution.** The hawksbill has a circumglobal distribution throughout tropical and, to a lesser extent, subtropical waters of the Atlantic, Indian, and Pacific Oceans. Satellite tagged turtles have shown significant variation in movement and migration patterns. In the Caribbean, distance traveled between nesting and foraging locations ranges from a few kilometers to a few hundred kilometers (Byles and Swimmer 1994; Hillis-Starr et al. 2000; Horrocks et al. 2001; Lagueux et al. 2003; Miller et al. 1998; Prieto et al. 2001). Only two hawksbill sea turtle sighting have been reported in the study area of the seismic surveys, with a few more west over the continental shelf (Belford et al. 2014).

**Migration and movement.** Upon first entering the sea, neonatal hawksbills in the Caribbean are believed to enter an oceanic phase that may involve long distance travel and eventual recruitment to nearshore foraging habitat (Boulon Jr. 1994). In the marine environment, the oceanic phase of juveniles (i.e., the "lost years") remains one of the most poorly understood aspects of hawksbill life history, both in terms of where turtles occur and how long they remain oceanic. Nesting site selection in the southwest Pacific appears to favor sites with higher wind and wave exposure, possibly as a means to aid hatchling dispersal (Garcon et al. 2010). Subadult hawksbill sea turtles satellite tracked in the Dry Tortugas National Park showed high-degrees of site fidelity for extended periods, although all three eventually moved to other areas outside the park (Hart et al. 2012). The same trend was found for adults tracked after nesting in the Dominican Republic, with some remaining for extended periods in the nesting area and other migrating to Honduras and Nicaragua (Hawkes et al. 2012). Satellite tracking for these individuals showed repeated returns to the same Dominican and Central American areas (Hawkes et al. 2012). Home ranges tend to be small (a few square kilometers)(Berube et al. 2012).

**Habitat.** Hawksbill sea turtles are highly migratory and use a wide range of broadly separated localities and habitats during their lifetimes (Musick and Limpus 1997; Plotkin 2003). Small juvenile hawksbills (5-21 cm straight carapace length) have been found in association with *Sargassum* spp. in both the Atlantic and Pacific Oceans (Musick and Limpus 1997) and observations of newly hatched hawksbills attracted to floating weed have been made (Hornell 1927; Mellgren and Mann 1996; Mellgren et al. 1994). Post-oceanic hawksbills may occupy a range of habitats that include coral reefs or other hard-bottom habitats, sea grass, algal beds, mangrove bays and creeks (Bjorndal and Bolten 2010; Musick and Limpus 1997), and mud flats (R. von Brandis, unpublished data in NMFS and USFWS 2007c). Eastern Pacific adult females have recently been tracked in saltwater mangrove forests along El Salvador and Honduras, a habitat that this species was not previously known to occupy (Gaos et al. 2011). Individuals of

multiple breeding locations can occupy the same foraging habitat (Bass 1999; Bowen et al. 1996; Bowen et al. 2007; Diaz-Fernandez et al. 1999; Velez-Zuazo et al. 2008). As larger juveniles, some individuals may associate with the same feeding locality for more than a decade, while others apparently migrate from one site to another (Blumenthal et al. 2009a; Mortimer et al. 2003; Musick and Limpus 1997). Larger individuals may prefer deeper habitats than their smaller counterparts (Blumenthal et al. 2009a). Nesting sites appear to be related to beaches with relatively high exposure to wind or wind-generated waves (Santana Garcon et al. 2010).

Within U.S. Caribbean territories and dependencies, hawksbill sea turtles nest principally in Puerto Rico and the U.S. Virgin Islands, particularly on Mona Island and Buck Island. They also nest on other beaches on St. Croix, Culebra Island, and Vieques Island, mainland Puerto Rico, St. John, and St. Thomas. Within the continental United States, hawksbill sea turtles nest only on beaches along the southeast coast of Florida and in the Florida Keys.

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

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

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

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

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

**Vocalization and hearing.** Sea turtles are low-frequency hearing specialists, typically hearing frequencies from 30 to 2,000 Hz, with a range of maximum sensitivity between 100 and 800 Hz (Bartol et al. 1999; Lenhardt 2002; Lenhardt 1994a; Moein Bartol and Ketten 2006; Ridgway et al. 1969). Piniak et al. (2012) found hawksbill hatchlings capable of hearing underwater sounds at frequencies of 50-1,600 Hz (maximum sensitivity at 200-400 Hz). Hearing below 80 Hz is less sensitive but still possible (Lenhardt 1994a).

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

**Status and trends.** Hawksbill sea turtles received protection on June 2, 1970 (35 FR 8495) under the Endangered Species Conservation Act and since 1973 have been listed as endangered under the ESA. Although no historical records of abundance are known, hawksbill sea turtles are considered to be severely depleted due to the fragmentation and low use of current nesting beaches (NMFS and USFWS 2007c). Consideration of the status of populations outside of the action area is important under the present analysis to determine the how risk the risk to the affected population(s) bears on the status of the species as a whole. Worldwide, an estimated 21,212-28,138 hawksbills nest each year among 83 sites. Among the 58 sites for with historic trends, all show a decline during the past 20 to 100 years. Among 42 sites for which recent trend data are available, 10 (24%) are increasing, three (7%) are stable and 29 (69%) are decreasing. Encouragingly, nesting range along Mexico and Central America appears not to have contracted (Gaos et al. 2010). Genetics supports roughly 6,000-9,000 adult females within the Caribbean (Leroux et al. 2012).

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

Population increase has been greater in the Insular Caribbean than along the Western Caribbean Mainland or the eastern Atlantic (including Sao Tomé and Equatorial Guinea). Nesting populations of Puerto Rico appeared to be in decline until the early 1990s, but have universally increased during the survey period. Mona Island now hosts 199-332 nesting females annually, and the other sites combined host 51-85 nesting females annually (R.P. van Dam and C.E. Diez, unpublished data in NMFS and USFWS 2007c)(C. E. Diez, Chelonia, Inc., in litt. to J. Mortimer 2006). At Buck Island Reef National Monument, protection has been in force since 1988, and during that time, hawksbill nesting has increased by 143% to 56 nesting females annually, with apparent spill over to beaches on adjacent St. Croix (Z. Hillis-Starr, National Park Service, in litt. to J. Mortimer 2006). However, St. John populations did not increase, perhaps due to the proximity of the legal turtle harvest in the British Virgin Islands (Z. Hillis-Starr, National Park Service, in litt. to J. Mortimer 2006). Populations have also been identified in Belize and Brazil as genetically unique (Hutchinson and Dutton 2007). An estimated 50-200 nests are laid per year in the Guinea-Bissau (Catry et al. 2009).

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

**Anthropogenic threats.** Threats to hawksbill sea turtles are largely anthropogenic, both historically and currently. Impacts to nesting beaches include the construction of buildings and pilings, beach armoring and renourishment, and sand extraction (Bouchard et al. 1998; Lutcavage et al. 1997b). Because hawksbills prefer to nest under vegetation (Horrocks and Scott 1991; Mortimer 1982), they are particularly impacted by beachfront development and clearing of dune vegetation (Mortimer and Donnelly in review). The presence of lights on or adjacent to nesting beaches alters the behavior of nesting adults (Witherington 1992) and is often fatal to emerging hatchlings as they are attracted to light sources and drawn away from the water (Witherington and Bjorndal 1991). One of the most detrimental human threats to hawksbill sea turtles is the intensive harvest of eggs from nesting beaches. Between 1950 and 1992, approximately 1.3 million hawksbill shells were collected to supply tortoiseshell to the Japanese market, the world's largest. Japan stopped importing tortoiseshell in 1993 in order to comply with Convention on the International Trade of Endangered Species (Limpus and Miller 2008). The U.S. Virgin Islands have a long history of tortoiseshell trade (Schmidt 1916).

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

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). 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 2010; Vargo et al. 1986a; Vargo et al. 1986b; 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 2010). 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 sea turtles (NOAA 2003).

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

**Critical habitat.** On September 2, 1998, the NMFS established critical habitat for hawksbill sea turtles around Mona and Monito Islands, Puerto Rico (63 FR 46693). Aspects of these areas that are important for hawksbill sea turtle survival and recovery include important natal development habitat, refuge from predation, shelter between foraging periods, and food for hawksbill sea turtle prey. No critical habitat occurs within the action area.

### 6.9 Kemp's ridley sea turtle

**Population.** Kemp's ridley sea turtles are considered to consist of a single population, although expansion of nesting may indicate differentiation.

**Distribution.** The Kemp's ridley was formerly known only from the Gulf of Mexico and along the Atlantic coast of the U.S. (TEWG 2000b). However, recent records support Kemp's ridley sea turtles distribution extending into the Mediterranean Sea on occasion (Tomas and Raga 2008). The vast majority of individuals stem from breeding beaches at Rancho Nuevo on the Gulf of Mexico coast of Mexico. Kemp's ridley sightings in the Mid-Atlantic Bight are largely over the continental shelf, with a few summer sightings over the continental shelf break near where seismic survey trackline (Belford et al. 2014).

**Movement and migration.** Tracking of post-nesting females from Rancho Nuevo and Texas beaches indicates that turtles move along coastal migratory corridors either to the north or south from the nesting beach (Byles 1989b; Byles and Plotkin 1994; Renaud 1995a; Renaud et al. 1996; Seney and Landry 2011; Shaver 1999; Shaver 2002) after remaining in the nesting area during the nesting period (Seney and Landry 2011). These migratory corridors appear to extend throughout the coastal areas of the Gulf of Mexico and most turtles appear to travel in waters less than roughly 50 m in depth. Turtles that headed north and east traveled as far as southwest Florida, whereas those that headed south and east traveled as far as the Yucatan Peninsula, Mexico (Morreale et al. 2007).

Kemp's ridleys in south Florida begin to migrate northward during spring. With each passing month, the waters to the north become warmer and turtles migrate further to Long Island Sound and even Nova Scotia in late summer (Bleakney 1955). During winter, individuals return south in response to local water temperatures; the turtles in the northernmost areas begin their southward movement first. By early November, turtles from New York and New Jersey merge with turtles from the Chesapeake Bay (Byles 1988; Keinath 1993; Lutcavage and Musick 1985; Renaud 1995a) and North Carolina inshore waters (Epperly et al. 1995a; Epperly et al. 1995b; Musick et al. 1994).

Following migration, Kemp’s ridley sea turtles settle into resident feeding areas for several months (Byles and Plotkin 1994; Morreale et al. 2007). Females may begin returning along relatively shallow migratory corridors toward the nesting beach in the winter in order to arrive at the nesting beach by early spring.

During spring and summer, juvenile Kemp’s ridleys occur in the shallow coastal waters of the northern Gulf of Mexico from south Texas to north Florida. In the fall, most Kemp’s ridleys migrate to deeper or more southern, warmer waters and remain there through the winter (Schmid 1998a). As adults, many turtles remain in the Gulf of Mexico, with only occasional occurrence in the Atlantic Ocean (NMFS et al. 2010). Satellite telemetry of males caught near Padre Island, Texas, indicates no migration, but year-round occurrence in nearshore waters less than 50 m deep (Shaver et al. 2005b). Many postnesting females from Rancho Nuevo migrate north to areas offshore of Texas and Louisiana (Marquez-M. 1994a). Farther south, some post-nesting females migrate from Rancho Nuevo to the northern and western Yucatán Peninsula in the southern Gulf of Mexico, which contains important seasonal foraging sites for adult females, such as the Bay of Campeche (Marquez-M. 1994a; Márquez 1990a; Pritchard and Marquez 1973).

**Reproduction.** Mating is believed to occur about three to four weeks prior to the first nesting (Rostal 2007), or late-March through early- to mid-April. It is presumed that most mating takes place near the nesting beach (Morreale et al. 2007; Rostal 2007). Females initially ovulate within a few days after successful mating and lay the first clutch approximately two to four weeks later; if a turtle nests more than once per season, subsequent ovulations occur within approximately 48 hours after each nesting (Rostal 2007).

Approximately 60% of Kemp's ridley nesting occurs along an 40 km stretch of beach near Rancho Nuevo, Tamaulipas, Mexico from April to July, with limited nesting to the north (100 nests along Texas in 2006) and south (several hundred nests near Tampico, Mexico in 2006 USFWS 2006). Nesting at this location may be particularly important because hatchlings can more easily migrate to foraging grounds (Putman et al. 2010). The Kemp's ridley sea turtle tends to nest in large aggregations or arribadas (Bernardo and Plotkin 2007). The period between Kemp's ridley arribadas averages approximately 25 days, but the precise timing of the arribadas is unpredictable (Bernardo and Plotkin 2007; Rostal et al. 1997). Like all sea turtles, Kemp's ridley sea turtles nest multiple times in a single nesting season. The most recent analysis suggests approximately 3.075 nests per nesting season per female (Rostal 2007). The annual average number of eggs per nest (clutch size) is 94 to 100 and eggs typically take 45 to 58 days to hatch, depending on temperatures (Marquez-M. 1994b; Rostal 2007; USFWS 2000; USFWS 2001; USFWS 2002; USFWS 2003; USFWS 2004; USFWS 2005; USFWS 2006). The period between nesting seasons for each female is approximately 1.8 to 2.0 years (Marquez et al. 1989; Rostal 2007; TEWG 2000b). The nesting beach at Rancho Nuevo may produce a "natural" hatchling sex ratio that is female-biased, which can potentially increase egg production as those turtles reach sexual maturity (Coyne and Landry Jr. 2007; Wibbels 2007).

**Growth.** Kemp's ridleys require approximately 1.5 to two (range 1-4) years to grow from a hatchling to a size of approximately 20 cm long, at which size they are capable of making a transition to a benthic coastal immature stage (Caillouet et al. 1995; Ogren 1989; Schmid 1998b; Schmid and Witzell 1997b; Snover et al. 2007b; TEWG 2000b; Zug et al. 1997). Based on the size of nesting females, it is assumed that turtles must attain a size of approximately 60 cm long prior to maturing (Marquez-M. 1994b). Growth models based on mark-recapture data suggest

that a time period of seven to nine years would be required for this growth from benthic immature to mature size (Schmid and Witzell 1997b; Snover et al. 2007b). Currently, age to sexual maturity is believed to range from approximately 10 to 17 years for Kemp's ridleys (Caillouet Jr. et al. 1995; Schmid and Witzell 1997a; Snover et al. 2007a; Snover et al. 2007b). However, estimates of 10 to 13 years predominate in previous studies (Caillouet et al. 1995; Schmid and Witzell 1997b; TEWG 2000b).

**Habitat.** Stranding data indicate that immature turtles in this benthic stage are found in coastal habitats of the entire Gulf of Mexico and U.S. Atlantic coast (Morreale et al. 2007; TEWG 2000b). Developmental habitats for juveniles occur throughout the entire coastal Gulf of Mexico and U.S. Atlantic coast northward to New England (Morreale et al. 2007; Schmid 1998b; Wibbels et al. 2005). Key foraging areas in the Gulf of Mexico include Sabine Pass, Texas; Caillou Bay and Calcasieu Pass, Louisiana; Big Gulley, Alabama; Cedar Keys, Florida; and Ten Thousand Islands, Florida (Carr and Caldwell 1956; Coyne et al. 1995; Ogren 1989; Schmid 1998b; Schmid et al. 2002; Witzell et al. 2005b). Foraging areas studied along the Atlantic coast include Pamlico Sound, Chesapeake Bay, Long Island Sound, Charleston Harbor, and Delaware Bay. Near-shore waters of 35 m or less provide the primary marine habitat for adults, although it is not uncommon for adults to venture into deeper waters (Byles 1989a; Mysing and Vanselous 1989; Renaud et al. 1996; Shaver et al. 2005a; Shaver and Wibbels 2007b).

Benthic coastal waters of Louisiana and Texas seem to be preferred foraging areas for Kemp’s ridley sea turtles (particularly passes and beachfronts), although individuals may travel along the entire coastal margin of the Gulf of Mexico (Landry and Costa 1999; Landry et al. 1996; Renaud 1995b). Sightings are less frequent during winter and spring, but this is likely due to lesser sighting effort during these times (Keinath et al. 1996; Shoop and Kenney 1992b).

**Feeding.** Kemp’s ridley diet consists mainly of swimming crabs, but may also include fish, jellyfish, and an array of mollusks. Immature Kemp’s ridleys off southwest Florida predate on benthic tunicates, a previously undocumented food source (Witzell and Schmid 2005).

**Diving.** Kemp’s ridley sea turtles can dive for well over 2.5 hours, although most dives are from 16 to 34 minutes (Mendonca and Pritchard 1986; Renaud 1995a). Individuals spend the vast majority of their time underwater; over 12-hour periods, 89% to 96% of their time is spent below the surface (Byles 1989b; Gitschlag 1996).

**Vocalization and hearing.** Sea turtles are low-frequency hearing specialists, typically hearing frequencies from 30 to 2,000 Hz, with a range of maximum sensitivity between 100 and 800 Hz (Bartol et al. 1999; Lenhardt 2002; Lenhardt 1994a; Moein Bartol and Ketten 2006; Ridgway et al. 1969). Hearing below 80 Hz is less sensitive but still possible (Lenhardt 1994a). Juvenile Kemp‘s ridleys can hear from 100 to 500 Hz, with a maximum sensitivity between 100 and 200 Hz at thresholds of 110 dB re 1 μPa (Moein Bartol and Ketten 2006).

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

**Status and trends.** The Kemp’s ridley sea turtle was listed as endangered on December 2, 1970 (35 FR 18319). Internationally, the Kemp’s ridley is considered the most endangered sea turtle

(NRC 1990a; USFWS 1999).

During the mid-20th century, the Kemp's ridley was abundant in the Gulf of Mexico. Historic information indicates that tens of thousands of Kemp's ridleys nested near Rancho Nuevo, Mexico, during the late 1940s (Hildebrand 1963). From 1978 through the 1980s, arribadas were 200 turtles or less, and by 1985, the total number of nests at Rancho Nuevo had dropped to approximately 740 for the entire nesting season, or a projection of roughly 234 turtles (TEWG 2000b; USFWS and NMFS 1992). Beginning in the 1990s, an increasing number of beaches in Mexico were being monitored for nesting, and the total number of nests on all beaches in Tamaulipas and Veracruz in 2002 was over 6,000; the rate of increase from 1985 ranged from 14-16% (Heppell et al. 2005; TEWG 2000b; USFWS 2002). In 2006, approximately 7,866 nests were laid at Rancho Nuevo with the total number of nests for all the beaches in Mexico estimated at about 12,000 nests, which amounted to about 4,000 nesting females based upon three nests per female per season (Rostal 2007; Rostal et al. 1997; USFWS 2006). Considering remigration rates, the population included approximately 7,000 to 8,000 adult female turtles at that time (Marquez et al. 1989; Rostal 2007; TEWG 2000b). The 2007 nesting season included an arribada of over 4,000 turtles over a three-day period at Rancho Nuevo (P. Burchfield, pers. comm. in NMFS and USFWS 2007b). The increased recruitment of new adults is illustrated in the proportion of first time nesters, which has increased from 6% in 1981 to 41% in 1994. Average population growth was estimated at 13% per year between 1991 and 1995 (TEWG 1998c). In 2008, there were 17,882 nests in Mexico (Gladys Porter Zoo 2008), and nesting in 2009 reached 21,144 (Burchfield 2010). In 2010, nesting declined significantly, to 13,302 but it is too early to determine if this is a one-time decline or if is indicative of a change in the trend. Preliminary estimates of 2011 and 2012 nesting supports 19,368 and 20,197 nests, respectively (back to 2009 levels)(Gallaway et al. 2013). Population modeling used by the TEWG (2000a) projected that Kemp's ridleys could reach the recovery plan's intermediate recovery goal of 10,000 nesters by the year 2015. Over one million hatchlings were released in 2011 and 2012 (Gallaway et al. 2013).

Nesting has also expanded geographically, with a Headstart program reestablishing nesting on South Padre Island starting in 1978. Growth remained slow until 1988, when rates of return started to grow slowly (Shaver and Wibbels 2007a). Nesting rose from 6 in 1996 to 128 in 2007, 195 in 2008, and 197 in 2009. Texas nesting then experienced a decline similar to that seen in Mexico for 2010, with 140 nests (National Park Service data, http://www.nps.gov/pais/naturescience/strp.htm), but nesting rebounded in 2011 with a record 199 nests (National Park Service data, http://www.nps.gov/pais/naturescience/current-season.htm).

Gallaway et al. (2013) estimated that nearly 189,000 female Kemp's ridley sea turtles over the age of two years were alive in 2012. Extrapolating based upon sex bias, the authors estimated that nearly a quarter million age two or older Kemp's ridleys were alive at this time.

**Natural threats.** Sea turtles face predation primarily by sharks and to a lesser extent by killer whales (Pitman and Dutton 2004). All sea turtles except leatherbacks can undergo "cold stunning" if water temperatures drop below a threshold level, which can pose lethal effects. Kemp's ridley sea turtles are particularly prone to this phenomenon along Cape Cod (Innis et al. 2009). From 2006-201), the number of cold-stunned turtles on Cape Cod beaches averaged 115 Kemp's ridleys. The fungal pathogens *Fusarium falciforme* and *F. keratoplasticum* an kill in

excess of 90% of sea turtle embryos they infect and may constitute a major threat to nesting productivity under some conditions (Sarmiento-Ramırez et al. 2014).

**Anthropogenic threats.** Population decline has been curtailed due to the virtual elimination of sea turtle and egg harvesting, as well as assistance in hatching and raising hatchlings (Headstart). However, habitat destruction remains a concern in the form of bottom trawling and shoreline development. Trawling destroys habitat utilized by Kemp's ridley sea turtles for feeding and construction activities can produce hazardous runoff. Bycatch is also a source of mortality for Kemp's ridley sea turtles (McClellan et al. 2009), with roughly three-quarters of annual mortality attributed to shrimp trawling prior to turtle excluder device (TED) regulations (Gallaway et al. 2013). However, this has dropped to an estimated one-quarter of total mortality nearly 20 years after TEDS were implemented in 1990 (Gallaway et al. 2013). In 2010, due to reductions in shrimping effort and TED use, shrimp-trawl related mortality appears to have dropped to 4% (1,884) of total mortality (65,505 individuals)(Gallaway et al. 2013). This increased to 3,300 individuals in 2012 (20% of total mortality)(Gallaway et al. 2013). Finkbeiner et al. (2011) estimated that annual bycatch interactions total at least 98,300 individuals annually for U.S. Atlantic fisheries (resulting in 2,700 mortalities or more). The vast majority of fisheries interactions with sea turtles in the U.S. are either Kemp's ridley's or loggerhead sea turtles (Finkbeiner et al. 2011).

Toxin burdens in Kemp's ridley sea turtles include DDT, DDE, PCBs, perfluorooctanoic acid (PFOA), perfluorooctanesulfonic acid (PFOS), chlordane, and other organochlorines (Keller et al. 2005; Keller et al. 2004a; Lake et al. 1994; Rybitski et al. 1995). These contaminants have the potential to cause deficiencies in endocrine, developmental and reproductive health, and are known to depress immune function in loggerhead sea turtles (Keller et al. 2006; Storelli et al. 2007b). Along with loggerheads, Kemp's ridley sea turtles have higher levels of PCB and DDT than leatherback and green sea turtles (Pugh and Becker 2001a). Organochlorines, including DDT, DDE, and PCBs have been identified as bioaccumulative agents and in greatest concentration in subcutaneous lipid tissue (Rybitski et al. 1995). Concentrations ranged from 7.46 mu g/kg to 607 mu g/kg, with a mean of 252 mu g/kg in lipid tissue. Five PCB congeners composed most of the contaminants: 153/132, 138/158, 180, 118, and 187 in order of concentration. PCBs have also been identified in the liver, ranging in concentration from 272 ng/g to 655 ng/g of wet weight, values that are several fold higher than in other sea turtle species (Lake et al. 1994). However, concentrations are reportedly 5% of that which causes reproductive failure in snapping turtles. DDE was identified to range from 137 ng/g to 386 ng/g wet weight. Trans-nonachlor was found at levels between 129 ng/g and 275 ng/g wet weight. Blood samples may be appropriate proxies for organochlorines in other body tissues (Keller et al. 2004a). Perfluorinated compounds in the forms of PFOA and PFOS have been identified in the blood of Kemp's ridley turtles at concentrations of 39.4 ng/mL and 3.57 ng/mL, respectively (Keller et al. 2005). Perfluorinated carboxylic acids (PFCAs) have also been detected. It is likely that age and habitat are linked to perflourinated chemical (PFC) bioaccumulation.

Oil can also be hazardous to Kemp's ridley turtles, with fresh oil causing significant mortality and morphological changes in hatchlings, but aged oil having no detectable effects (Fritts and McGehee 1981). 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). 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 2010; Vargo et al. 1986a; Vargo et al. 1986b; 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 2010). 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).

Blood levels of metals are lower in Kemp's ridley sea turtles than in other sea turtles species or similar to them, with copper (215 ng/g to 1,300 ng/g), lead (0 to 34.3 ng/g), mercury (0.5 ng/g to 67.3 ng/g), silver (0.042 ng/g to 2.74 ng/g), and zinc (3,280 ng/g to 18,900 ng/g) having been identified (Innis et al. 2008; Orvik 1997). It is likely that blood samples can be used as an indicator of metal concentration. Mercury has been identified in all turtle species studied, but are generally an order of magnitude lower than toothed whales. The higher level of contaminants found in Kemp's ridley sea turtles are likely due to this species tendency to feed higher on the food chain than other sea turtles. Females from sexual maturity through reproductive life should have lower levels of contaminants than males because contaminants are shared with progeny through egg formation.

**Critical habitat.** NMFS has not designated critical habitat for Kemp's ridley sea turtle.

### 6.10 Leatherback sea turtle

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

**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 2006; Márquez 1990b; Spotila et al. 1996).

**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 1990b; Threlfall 1978). Pacific ranges extend to Alaska, Chile, and New Zealand (Brito 1998; Gill 1997; Hodge and Wing 2000). About 10o leatherback sightings have occurred in the area near the seismic survey, with hundreds of others in waters surrounding it, all mostly during spring, summer, or fall (most common in summer) (Belford et al. 2014). Sightings are most common over the continental shelf to the shelf break, but sightings in deeper water are also frequent (Belford et al. 2014).

Leatherbacks also occur in Mediterranean and Indian Oceans (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 1982c). 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).

**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 (Rhodin 1985) or 13-14 years (Zug and Parham 1996). However, recent research suggests otherwise, with western North Atlantic leatherbacks possibly not maturing until as late as 29 years of age (Avens and Goshe 2007; Avens and Goshe 2008; Avens et al. 2009). Female leatherbacks nest frequently (up to 13, average of 5-7 nests per year and about every 2-3 years)(Eckert et al. 2012). The average number of eggs per clutch varies by region: Atlantic Ocean (85 eggs), western Pacific Ocean (85 eggs), eastern Pacific Ocean (65 eggs) and Indian Ocean (>100 eggs (Eckert et al. 2012)). However, up to ~30% 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 1992a; Starbird et al. 1993). Movements are largely dependent upon reproductive and feeding cycles and the oceanographic features that concentrate prey, such as frontal systems, eddy features, current boundaries, and coastal retention areas (Benson et al. 2011b; Collard 1990; Davenport and Balazs 1991; Frazier 2001; HDLNR 2002). Aerial surveys off the western U.S. support continental slope waters as having greater leatherback occurrence than shelf waters (Bowlby et al. 1994; Carretta and Forney 1993; Green et al. 1992; Green et al. 1993). Nesting sites appear to be related to beaches with relatively high exposure to wind or wind-generated waves (Santana Garcon et al. 2010).

Areas above 30º N in the Atlantic appear to be popular foraging locations (Fossette et al. 2009b). Northern foraging areas were proposed for waters between 35º and 50º N along North American, Nova Scotia, the Gulf of Saint-Laurent, in the western and northern Gulf Stream, the Northeast Atlantic, the Azores front and northeast of the Azores Islands, north of the Canary Islands.

Southern foraging was proposed to occur between 5º and 15º N in the Mauritania upwelling, south of the Cape Verde islands, over the Guinea Dome area, and off Venezuela, Guyana and Suriname.

**Migration and movement.** Leatherback sea turtles migrate throughout open ocean convergence zones and upwelling areas, along continental margins, and in archipelagic waters (Eckert 1998; Eckert 1999; Morreale et al. 1994). In a single year, a leatherback may swim more than 11,000 km to nesting and foraging areas throughout ocean basins (Benson et al. 2007a; Benson et al. 2011b; 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 (generally within 100-300 km)(Benson et al. 2011a; Eckert et al. 2012), or range widely, presumably to feed on available prey (Byrne et al. 2009; Fossette et al. 2009a).

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

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

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

**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. 2005a).

**Feeding.** Leatherbacks may forage in high-invertebrate prey density areas formed by favorable oceanographic features (Eckert 2006; Ferraroli et al. 2004). Although leatherbacks forage in coastal waters, they appear to remain primarily pelagic through all life stages (Heppell et al. 2003). The location and abundance of prey, including medusae, siphonophores, and salpae, in temperate and boreal latitudes likely has a strong influence on leatherback distribution in these areas (Plotkin 1995).

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

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

**Status and trends.** Leatherback sea turtles received protection on June 2, 1970 (35 FR 8491) under the Endangered Species Conservation Act and, since 1973, have been listed as endangered

under the ESA, but declines in nesting have continued worldwide. Consideration of the status of populations outside of the action area is important under the present analysis to determine the how risk the risk to the affected population(s) bears on the status of the species as a whole. Breeding females were initially estimated at 29,000-40,000, 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 2004b). The species as a whole is declining and local populations are in danger of extinction (NMFS 2001b; NMFS 2001a)(Table 9).

**Table 9.** Leatherback nesting population site location information where multiple-year surveys were conducted or trends are known (data type, years surveyed, annual number (nests, females, trend). Nesting population trend symbols: ▲ = increasing; ▼ = decreasing; ▬ = stable; ? = unknown.

| Location | Data: Nests, Females | Years | Annual number | Trend | Reference |
|---|---|---|---|---|---|
| **Atlantic** | | | | | |
| United States (Florida) | Nests | 1979 - 2008 | 63-754 | ▲ | Stewart et al. (2011) |
| Puerto Rico (Culebra) | Nests | 1993 - 2012 | 395-32 | ▼ | {C. Diez, Department of Natural and Environmental Resources of Puerto Rico, unpublished data in NMFS and USFWS, 2013 #36241} Diez et al. (2010; Ramírez-Gallego et al. 2013) |
| Puerto Rico (other) | Nests | 1993 - 2012 | 131-1,291 | ▲ | C. Diez, Department of Natural and Environmental Resources of Puerto Rico, unpublished data in NMFS and USFWS (2013) |
| United States Virgin Islands (Sandy Point National Wildlife Refuge, St. Croix) | Nests | 1986 - 2004 | 143-1,008 | ▲[1] | Dutton et. al. (2005); Turtle Expert Working Group (2007b) |
| British Virgin Islands | Nests | 1986 - 2006 | 0-65 | ▲ | McGowan et al. (2008) ;Turtle Expert Working Group (2007b) |
| Nicaragua | Nests | 2008 - 2013 | 42-132 | ?[2] | {C. Laguex and C. Campbell, Wildlife Conservation Society, unpublished data in NMFS and USFWS, 2013 #36241} |
| Costa Rica (Tortuguero) | Nests | 2007 - 2011 | ~281 | ▼ | Gordon and Harrison (2012) |
| Costa Rica (Gandoca) | Nests | 1990 - 2004 | ~583 | ▼ | Chacón and Eckert (2007); Turtle Expert Working Group (2007b) |
| Panama (Chiriqui Beach) | Nests | 2004 | 1,000- | ? | Meylan et al. (2013) |

| Location | Data: Nests, Females | Years | Annual number | Trend | Reference |
|---|---|---|---|---|---|
| | | - 2011 | 4,999 | | |
| Colombia | Nests | 2006 - 2007 | 1,653-2,871 | ? | Patino-Martinez et al. (2008) |
| Trinidad | Females | 1994 - 2005 | 2,096 | ▲ | Turtle Expert Working Group (2007b) |
| Guyana | Nests | 2007 - 2010 | 377-1,722 | ▲ | De Freitas and Pritchard (2008; 2009; 2010); Turtle Expert Working Group (2007b); Kalamandeen et al. (2007) |
| French Guiana | Nests | | 5,029-63,294 | — | Fossette et al. (2008) |
| Suriname | Nests | | 2,732-31,000 | — | Fossette et al. (2008) |
| Brazil | Nests | 1988 - 2004 | 6-527 | ▲ | Thomé et al. (Thomé et al. 2007); Turtle Expert Working Group (2007b) |
| Equatorial Guinea (Bioko) | Nests | 2000 - 2005 | 2,127-5,071 | ? | Rader et al. (2006) |
| Congo | Nests | 2003 - 2006 | 70-148 | ? | Rentaura (2004; 2006) |
| Gabon | Nests | 2002 - 2007 | 36,185-126,480 | ? | Witt et al. (2009) |

[1] A more recent trend analysis was not found in the literature. However, trends since 2001 suggest the population may be declining, possibly due to a decrease in the number of new nesters, lowered productivity (number of clutches per season and lower hatch success), and an increase in remigration intervals (Garner 2012; Garner et al. 2012).

[2] The number of nests likely underrepresents the area because 22% of nesting activity was not surveyed from 2011-2013 due to military presence {Laguex and Campbell, Wildlife Conservation Society, unpublished data in NMFS and USFWS, 2013 #36241}.

[3] Based on 12.8 km index area in Maputaland and St. Lucia Marine Reserves, South Africa.

[4] Survey distance and time differed between the two surveys at Labu Tali, but the weight of evidence from the area indicates a declining population.

Nesting aggregations occur along Gabon, Sao Tome and Principe, French Guiana, Suriname, and Florida (Bräutigam and Eckert 2006; Márquez 1990b; 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 2001a). 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 2007c). Nesting in Culebra, Puerto Rico has declined since 2004, has slowed in the U.S. Virgin Islands from 2001-2010, and increased by 10% annually in Florida from 1979-2008 (NMFS USFWS 2013).

The largest nesting aggregation in the western North Atlantic occurs in French Guiana and Suriname and likely belongs to a metapopulation whose limits remain unknown (Rivalan et al. 2006). For Suriname and French Guiana, historical estimates of the number of females nesting each year range from approximately 5,000 to 20,000 (Fossette et al. 2008). Suriname and French Guiana may represent over 40% of the world's leatherback population, although the magnitude of the West African rookery needs to be verified (Spotila et al. 1996). Heppell et al. (2003a) concluded that leatherbacks generally show less genetic structuring than green and hawksbill sea turtles. The French Guiana nesting aggregation has declined ~15% annually since 1987 (NMFS 2001a). However, from 1979-1986, the number of nests increased ~15% annually, possibly indicating the current decline may be linked with the erosion cycle of Guiana beaches (NMFS 2006e). Girondot et al. (2007a) analyzed nesting data collected between 1967 and 2002 from French Guiana and Suriname and found that the population can be classified as stable or slightly increasing. The Turtle Expert Working Group (2007b) analyzed nest numbers from 1967-2005 and found a positive population growth rate over the 39-year period for French Guiana and Suriname. 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. 2007b; 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 2007c). Trinidad supports an estimated 7,000 to 12,000 leatherbacks nesting annually (Stewart et al. 2013), which represents more than 80% of the nesting in the insular Caribbean Sea (Fournillier and Eckert 1999). Using both Bayesian modeling and regression analyses, the TEWG (2007c) 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 2007c). 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 2007c). 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 2007c). Overall increases are recorded for mainland Puerto Rico and St. Croix, as well as the U.S. Virgin Islands (Ramírez-Gallego et al. 2013). Trends since 2001 suggest the population may be declining, possibly due to a decrease in the number of new nesters, lowered productivity (number of clutches per season and lower hatch success), and an increase in remigration intervals (Garner 2012; Garner et al. 2012).

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 (2007c) 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 2007c).

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

**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). The fungal pathogens *Fusarium falciforme* and *F. keratoplasticum* an kill in excess of 90% of sea turtle embryos they infect and may constitute a major threat to nesting productivity under some conditions (Sarmiento-Ramırez et al. 2014).

**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 are often fatal to emerging hatchlings as they are drawn to light sources and away from the sea (Bourgeois et al. 2009; Cowan et al. 2002; Deem et al. 2007; Witherington 1992; Witherington and Bjorndal 1991). Leatherbacks are much more likely to emerge and not nest on developed beaches and much more likely to emerge and nest on undeveloped stretches (Roe et al. 2013). One study found 37% of dead leatherback turtles had ingested various types of plastic 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). A leatherback found stranded along the northern Adriatic had been weakened by plastic ingestion, likely leading to an infection that ultimately killed the individual (Poppi et al. 2012). Although global warming may expand foraging habitats into higher latitude waters, increasing temperatures may increase feminization of nests (Hawkes et al. 2007b; James et al. 2006; McMahon and Hays 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. Observer coverage for this period ranged from 54 to 92%. Trinidad and Tobago's Institute for Marine Affairs estimated that more than 3,000 leatherbacks were captured incidental to gillnet fishing in the coastal waters of Trinidad in 2000. Half or more of the gravid turtles in Trinidad and Tobago waters may be killed (Lee Lum 2003), though many of the turtles do not die as a result of drowning, but rather because the fishermen butcher them in order to get them out of their nets (NMFS 2001b).

Leatherback sea turtles are known to drown in fish nets set in coastal waters of Sao Tome, West Africa (Castroviejo et al. 1994; Graff 1995). Gillnets are one of the suspected causes for the decline in the leatherback turtle population in French Guiana (Chevalier et al. 1999), and gillnets targeting green and hawksbill turtles in the waters of coastal Nicaragua also incidentally catch

leatherback turtles (Lagueux 1998). Observers on shrimp trawlers operating in the northeastern region of Venezuela documented the capture of six leatherbacks from 13,600 trawls (Marcano and Alió-M 2000). An estimated 1,000 mature female leatherback turtles are caught annually off of Trinidad and Tobago with mortality estimated to be between 50-95% (Eckert and Lien 1999). There are known to be many sizeable populations of leatherbacks nesting in West Africa, possibly as many as 20,000 females nesting annually (Fretey 2001b). In Ghana, nearly two thirds of the leatherback turtles that come up to nest on the beach are killed by local fishermen.

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). 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 2010; Vargo et al. 1986a; Vargo et al. 1986b; 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 2010).

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). Along with these, lead has also been reported in high concentrations, potentially to the detriment of the individual (Perrault et al. 2013; Poppi et al. 2012). 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. Virgin Islands 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. This critical habitat does not co-occur with the action area.

### 6.11 Loggerhead sea turtle- Northwest Atlantic DPS

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

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.

**Distribution.** Loggerheads are circumglobal, occurring throughout the temperate and tropical regions of the Atlantic Ocean. Loggerheads are the most abundant species of sea turtle found in U.S. coastal waters. Loggerheads are sighted more frequently in the region than any other sea turtle species (Belford et al. 2014). Sightings are concentrated over the continental shelf, but are routine east over the shelf break and into deeper waters, particularly in summer (Belford et al. 2014). Hundreds of occurrences have been documented in the study area (Belford et al. 2014).

**Reproduction and growth.** Loggerhead nesting is confined to lower latitude 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 (Avens et al. 2013; NMFS 2005a). Loggerheads in the Mediterranean, similar to those in the Atlantic, grow at roughly 11.8 cm/yr for the first six months and slow to roughly 3.6 cm/yr at age 2.5-3.5. As adults, individuals may experience a secondary growth pulse associated with shifting into neritic habitats, although growth is generally monotypic (declines with age Casale et al. 2009a; Casale et al. 2009b). Individually-based variables likely have a high impact on individual 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 1985a; Frazer et al. 1994; NMFS 2001b; Witherington et al. 2006). However, based on data from tag returns, strandings, and nesting surveys, NMFS (2001b) estimated ages of maturity ranging from 20-38 years and benthic immature stage lasting from 14-32 years. Notably, data from several studies showed decreased growth rates of loggerheads in U.S. Atlantic waters from 1997-2007, corresponding to a period of 43% decline in Florida nest counts (Bjorndal et al. 2013).

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

generated waves (Santana Garcon et al. 2010).

Nesting in the Gulf of Mexico does occur, although primarily in Florida, with rare nests along North and South Padre Island in Texas (Dodd 1988b; Hildebrand 1983).

**Migration and movement.** Loggerhead hatchlings migrate offshore and become associated with *Sargassum* spp. habitats, driftlines, and other convergence zones (Carr 1986). After 14-32 years of age, they shift to a benthic habitat, where immature individuals forage in the open ocean and coastal areas along continental shelves, bays, lagoons, and estuaries (Bowen et al. 2004; NMFS 2001b). Adult loggerheads make lengthy migrations from nesting beaches to foraging grounds (TEWG 1998a). In the Gulf of Mexico, larger females tend to disperse more broadly after nesting than smaller individuals, which tend to stay closer to their nesting locations (Girard et al. 2009). In the North Atlantic, loggerheads travel north during spring and summer as water temperatures warm and return south in fall and winter, but occur offshore year-round assuming adequate temperature. As water temperatures drop from October to December, most loggerheads emigrate from their summer developmental habitats to warmer waters south of Cape Hatteras, where they winter (Morreale and Standora 1998). For immature individuals, this movement occurs in two patterns: a north-south movement over the continental shelf with migration south of Cape Hatteras in winter and movement north along Virginia for summer foraging, and a not-so-seasonal oceanic dispersal into the Gulf Stream as far north as the 10-15˚ C isotherm (Mansfield et al. 2009). Wallace et al. (2009) suggested differences in growth rate based upon these foraging strategies. Long Island Sound, Core Sound, Pamlico Sound, Cape Cod Bay, and Chesapeake Bay are the most frequently used juvenile developmental habitats along the Northeast United States Continental Shelf Large Marine Ecosystem (Burke et al. 1991; Epperly et al. 1995a; Epperly et al. 1995b; Epperly et al. 1995c; Mansfield 2006; Prescott 2000; University of Delaware Sea Grant 2000). 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 α levels lower. Satellite tracking of loggerheads from southeastern U.S. nesting beaches supports three dispersal modes to foraging areas: one northward along the continental shelf to the northeastern U.S., broad movement through the southeastern and mid-Atlantic U.S., and residency near breeding areas (Reina et al. 2012).

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

**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 (Dodd 1988a; NMFS 2001b; Rees and Margaritoulis 2004). Nest temperature seems to drive sex determination. Along Florida, males primarily derive from earlier-season nests (LeBlanc et al. 2012). Here, nests ranged from an average sex ratio of 55% female to 85% (LeBlanc et al. 2012).

Additionally, little is known about longevity, although Dodd (1988a) estimated the maximum female life span at 47-62 years. Heppell et al. (2003a) 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). Another recent estimate suggested a survival rate of 0.41 or 0.60 (C.I.s 0.20-0.65 and 0.40-0.78, respectively), depending upon assumptions within the study (Sasso et al. 2011). Survival rates for hatchlings during their first year are likely very low (Heppell et al. 2003a; Heppell et al. 2003).

**Feeding.** Loggerhead sea turtles are omnivorous and opportunistic feeders through their lifetimes (Parker et al. 2005). Hatchling loggerheads feed on macroplankton associated with *Sargassum* spp. communities (NMFS and USFWS 1991b). Pelagic and benthic juveniles forage on crabs, mollusks, jellyfish, and vegetation at or near the surface (Dodd 1988a; Wallace et al. 2009). Loggerheads in the deep, offshore waters of the western North Pacific feed on jellyfish, salps, and other gelatinous animals (Dodd Jr. 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 five 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 2004a). During these dives, there were also several strong surface temperature fronts that individuals were associated with, one of 20° C at 28° N latitude and another of 17° C at 32° N latitude. In the Mediterranean, dives of over 300 min have been recorded in association with depressed water temperatures and are proposed as an overwintering strategy (Luschi et al. 2013).

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

These hearing sensitivities are similar to those reported for two terrestrial species: pond and wood turtles. Pond turtles respond best to sounds between 200 and 700 Hz, with slow declines

below 100 Hz and rapid declines above 700 Hz, and almost no sensitivity above 3 kHz (Wever and Vernon 1956). Wood turtles are sensitive up to about 500 Hz, followed by a rapid decline above 1 kHz and almost no responses beyond 3 or 4 kHz (Patterson 1966).

**Status and trends.** Loggerhead sea turtles were listed as threatened under the ESA of 1973 on July 28, 1978 (43 FR 32800). On September 22, 2011, the NMFS designated nine distinct population segments (DPSs) of loggerhead sea turtles (75 FR 12598).

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 2004b).

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, and South Africa (EuroTurtle 2006 as cited in LGL Ltd. 2007; Márquez 1990b).

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 are 32,000-56,000. All of these are currently in decline or data are insufficient to assess trends (NMFS 2001b; TEWG 1998c). 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).

The northern recovery unit along Georgia, South Carolina, and North Carolina has a forty-year time-series trend showing an overall decline in nesting, but the shorter comprehensive survey data (20 years) indicate a stable population (Georgia Department of Natural Resources, North Carolina Wildlife Resources Commission, and South Carolina Department of Natural Resources nesting data located at www.seaturtle.org). NMFS scientists have estimated that the northern subpopulation produces 65% males (NMFS 2001b).

The peninsular Florida recovery unit is the largest loggerhead nesting assemblage in the northwest Atlantic. A near-complete nest census (all beaches including index nesting beaches) undertaken from 1989 to 2007 showed a mean of 64,513 loggerhead nests per year, representing approximately 15,735 nesting females annually (NMFS and USFWS 2008). The statewide estimated total for 2010 was 73,702 (FWRI nesting database). An analysis of index nesting beach data shows a 26% nesting decline between 1989 and 2008, and a mean annual rate of decline of 1.6% despite a large increase in nesting for 2008, to 38,643 nests (FWRI nesting database)(NMFS and USFWS 2008; Witherington et al. 2009). In 2009, nesting levels, while still higher than the lows of 2004, 2006, and 2007, dropped below 2008 levels to approximately

32,717 nests, but in 2010, a large increase was seen, with 47,880 nests on the index nesting beaches (FWRI nesting database). The 2010 index nesting number is the largest since 2000. With the addition of data through 2010, the nesting trend for the northwestern Atlantic DPS is slightly negative and not statistically different from zero (no trend)(NMFS and USFWS 2010).

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[3]. Loggerhead nesting is thought to consist of just 60 nesting females in the Caribbean and Gulf of Mexico (NMFS 2006c). 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.

Zurita et al. (2003) found a statistically significant increase in the number of nests on seven of the beaches on Quintana Roo, Mexico, from 1987-2001, where survey effort was consistent during the period. However, nesting has declined since 2001, and the previously reported increasing trend appears to have been temporary (NMFS and USFWS 2008).

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

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3 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.

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

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. Shrimp trawl fisheries account for the highest number of captured and killed loggerhead sea turtles. Pacific bycatch is about 400 individuals annually in U.S. fisheries resulting in at least 20 mortalities (Finkbeiner et al. 2011). Each year, various fisheries capture about 2,000 loggerhead sea turtles in Pamlico Sound, of which almost 700 die. As a result of the 2006 and 2007 tri-national fishermen's exchanges in 2007 a prominent Baja California Sur fleet retired its bottom-set longlines (Peckham and Maldonado-Diaz 2012; Peckham et al. 2008). Prior to this closure, the longline fleet interacted with an estimated 1,160-2,174 loggerheads annually, with nearly all (89%) of the takes resulting in mortalities (Peckham et al. 2008). Offshore longline tuna and swordfish longline fisheries are also a serious concern for the survival and recovery of loggerhead sea turtles and appear to affect the largest individuals more than younger age classes (Aguilar et al. 1995; Bolten et al. 1994; Carruthers et al. 2009; Howell et al. 2008; Marshall et al. 2009; Petersen et al. 2009; Tomás et al. 2008).

Marine debris ingestion is 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). 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). Another study in the Tyrrhenian Sea found 71% of stranded and bycaught sea turtles had plastic debris in their guts (Campani et al. 2013). Another threat marine debris poses is to hatchlings on beaches escaping to the sea. Two thirds of loggerheads contacted marine debris on their way to the ocean and many became severely entangled or entrapped by it (Triessnig et al. 2012).

Climate change may also have significant implications on loggerhead populations worldwide. In addition to potential loss of nesting habitat due to sea level rise, loggerhead sea turtles are very sensitive to temperature as a determinant of sex while incubating. Ambient temperature increase by just 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. 2008c). Warmer temperatures may also decrease the energy needs of a developing embryo (Reid et al. 2009). Pike (2014) estimated that loggerhead populations in tropical areas produce about 30% fewer hatchlings than do populations in temperate areas. Historical climactic patterns have been attributed to the decline in loggerhead nesting in Florida, but evidence for this is tenuous (Reina et al. 2013).

Tissues taken from loggerheads sometimes contain very high levels of organochlorines chlorobiphenyl, chlordanes, lindane, endrin, endosulfan, dieldrin, PFOS, PFOA, DDT, and PCB (Alava et al. 2006; Corsolini et al. 2000; Gardner et al. 2003; Guerranti et al. 2013; Keller et al. 2005; Keller et al. 2004a; Keller et al. 2004b; McKenzie et al. 1999; Monagas et al. 2008; Oros et al. 2009; Perugini et al. 2006; Rybitski et al. 1995; Storelli et al. 2007a). It appears that levels of organochlorines have the potential to suppress the immune system of loggerhead sea turtles and may affect metabolic regulation (Keller et al. 2004c; Keller et al. 2006; Oros et al. 2009). These contaminants could cause deficiencies in endocrine, developmental, and reproductive health (Storelli et al. 2007a). It is likely that the omnivorous nature of loggerheads makes them more prone to bioaccumulating toxins than other sea turtle species (Godley et al. 1999; McKenzie et al. 1999).

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). 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 2010; Vargo et al. 1986a; Vargo et al. 1986b; Vargo et al. 1986c). PAH pollution from petroleum origins has been found in Cape Verde loggerheads, where marine oil and gas extraction is not undertaken (Camacho et al. 2012). 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 2010). Oil can also cause indirect effects to sea turtles through impacts to habitat and prey organisms. The loss of invertebrate communities due to oiling or oil toxicity would also decrease prey availability for loggerhead sea turtles (NOAA 2003). Furthermore, 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).

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 pass to turtles 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 2001b). Arsenic occurs at levels several fold more concentrated in loggerhead sea turtles than marine mammals or seabirds.

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

**Critical habitat.** On July 10, 2014, NMFS finalized a rule designating critical habitat for Northwest Atlantic Ocean DPS loggerhead sea turtles (79 FR 39855). This includes a *Sargassum* habitat area that co-occurs with the proposed seismic surveys in offshore areas.

# 7 ENVIRONMENTAL BASELINE

By regulation, the environmental baseline for ESA section 7 consultation includes 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.

## 7.1 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 2002; IPCC 2014). 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 as well as an increase in the mass of the Antarctic and Greenland ice sheets, 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 ESA-listed species to rise with the degree of climate shift associated with global warming.

Indirect effects of climate change would result from changes in the distribution of temperatures suitable for whale calving and rearing, the distribution and abundance of prey, and abundance of competitors or predators. For species that undergo long migrations, individual movements are usually associated 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). With warming temperatures and decreasing sea ice, humpback and fin whales have been found in increasing numbers at the northern extreme of their Pacific range and are regularly found now in the southern Chukchi Sea (Clarke et al. 2013). We do not know if this is due to range expansion owing to species recovery, or due to altered habitat associated with climate change (Clarke et al. 2013). Climate change can influence reproductive success by altering prey availability, as evidenced by high success of northern elephant seals during El Niño periods, when cooler, more productive waters are associated with higher first year pup survival (McMahon and Burton. 2005).

Reduced prey availability resulting from increased sea temperatures has also been suggested to explain reductions in Antarctic fur seal pup and harbor porpoise survival (Forcada et al. 2005; Macleod et al. 2007). Polygamous marine mammal mating systems can also be perturbed by rainfall levels, with the most competitive grey seal males being more successful in wetter years than in drier ones (Twiss et al. 2007). Sperm whale females were observed to have lower rates of conception following unusually warm sea surface temperature periods (Whitehead 1997). Marine mammals with restricted distributions linked to water temperature may be particularly exposed to range restriction (Issac 2009; Learmonth et al. 2006). MacLeod (2009) estimated that, based upon expected shifts in water temperature, 88% of cetaceans would be affected by climate change, 47% would be negatively affected, and 21% would be put at risk of extinction. Of greatest concern are cetaceans with ranges limited to non-tropical waters and preferences for shelf habitats (Macleod 2009). Modeling of North Atlantic cetacean species found that three of four odontocete species would likely undergo range contraction while one would expand its range (Lambert et al. 2014). Kaschner et al. (2011) modeled marine mammal species richness, overlaid with projections of climate change and found that species in lower-latitude areas would likely be more affected than those in higher-latitude regions. Variations in the recruitment of krill and the reproductive success of krill predators correlate to variations in sea-surface temperatures and the extent of sea-ice cover during winter months. Although the Intergovernmental Panel on Climate Change (2001) did not detect significant changes in the extent of Antarctic sea-ice using satellite measurements, Curran et al. (2003) analyzed ice-core samples from 1841 to 1995 and concluded Antarctic sea ice cover had declined by about 20% since the 1950s.

Roughly 50% of the Earth's marine mammal biomass occurs in the Southern Ocean, with all baleen whales feeding largely on a single krill species, *Euphausia superba*, here and feeding virtually nowhere else (Boyd 2002). However, Atkinson et al. (2004) found severe decreases in krill populations over the past several decades in some areas of the Antarctic, linked to sea ice loss. Reid and Croxall (2001) analyzed a 23-year time series of the reproductive performance of predators (Antarctic fur seals, gentoo penguins, macaroni penguins, and black-browed albatrosses) that depend on krill for prey and concluded that these populations experienced increases in the 1980s followed by significant declines in the 1990s accompanied by an increase in the frequency of years with reduced reproductive success. The authors concluded that macaroni penguins and black-browed albatrosses had declined by as much as 50% in the 1990s, although incidental mortalities from longline fisheries probably contributed to the decline of the

albatross. However, these declines resulted, at least in part, from changes in the structure of the krill population, particularly reduced recruitment into older krill age classes, which lowered the number of predators krill could sustain. The authors concluded that the biomass of krill within the largest size class was sufficient to support predator demand in the 1980s but not in the 1990s. By 2055, severe reductions in fisheries catch due to climate change have been suggested to occur in the Indo-Pacific, Red Sea, Mediterranean Sea, Antarctic, and tropical areas worldwide while increased catches are expected in the Arctic, North Pacific, North Atlantic, and northern portions of the Southern Ocean (Cheung et al. 2010).

Climate-mediated changes in the distribution and abundance of keystone prey species like krill and climate-mediated changes in the distribution of cephalopod populations worldwide is likely to affect marine mammal populations as they re-distribute throughout the world's oceans in search of prey. If sea ice extent decreases, then larval krill may not be able to survive without access to underice algae to feed on. This may be a cause of decreased krill abundance in the northwestern Antarctic Peninsula during the last decade (Fraser and Hofmann 2003). Meltwaters have also reduced surface water salinities, shifting primary production along the Antarctic Peninsula (Moline et al. 2004). Blue whales, as predators that specialize in eating krill, are likely to change their distribution in response to changes in the distribution of krill (Clapham et al. 1999; Payne et al. 1986; Payne et al. 1990b). If they did not change their distribution or could not find the biomass of krill necessary to sustain their population numbers, their populations would likely experience declines similar to those observed in other krill predators, including dramatic declines in population size and increased year-to year variation in population size and demographics. These outcomes would dramatically increase the extinction probability of baleen whales. Edwards et al. (2007) found a 70% decrease in one zooplankton species in the North Sea and an overall reduction in plankton biomass as warm-water species invade formerly cold-water areas. However, in other areas, productivity may increase, providing more resources for local species (Brown et al. 2009). This has been proposed to be the case in the eastern North Pacific, where a poleward shift in the North Pacific Current that would likely continue under global warming conditions would enhance nutrient and planktonic species availability, providing more prey for many higher trophic level species (Sydeman et al. 2011). Species such as gray whales may experience benefits from such a situation (Salvadeo et al. 2013). In addition, reductions in sea ice may alleviate "choke points" that allow some marine mammals to exploit additional habitats (Higdon and Ferguson 2009). Similar scenarios may play out in the action area.

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 (Schumann et al. 2013; 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).

Sims et al. (2001) found the timing of squid peak abundance in the English Channel advanced by

120-150 days in the warmest years compared with the coldest. Bottom water temperatures correlated with the extent of squid movement, and temperature increases over the five months before and during the month of peak squid movement did not differ between early and late years. These authors concluded that the temporal variation in peak abundance of squid seen off Plymouth represents temperature-dependent movement, which climatic changes association with the North Atlantic Oscillation mediate. Cephalopods dominate the diet of sperm whales, who would likely re-distribute following changes in the distribution and abundance of their prey. If, however, cephalopod populations collapse or decline dramatically, sperm whales would likely decline as well. Long-term shifts of sperm whale prey in the California Current have also been attributed to the re-distribution of their prey resulting from climate-based shifts in oceanographic variables (Salvadeo et al. 2011). Similar changes have also been suggested for sardines and anchovy in the California Current (Salvadeo et al. 2011), which are important prey for humpback and fin whales, among others.

Climate change has been linked to changing ocean currents as well. Rising carbon dioxide levels have been identified as a reason for a poleward shift in the Eastern Australian Current, shifting warm waters into the Tasman Sea and altering biotic features of the area (Johnson et al. 2011; Poloczanska et al. 2009). Similarly, the Kuroshio Current in the western North Pacific (an important foraging area for juvenile sea turtles) has shifted southward as a result of altered long-term wind patterns over the Pacific Ocean (Poloczanska et al. 2009). Ocean temperatures around Iceland are linked with alterations in the continental shelf ecosystem there, including shifts in minke whale diet (Víkingsson et al. 2014).

Apart from species-specific impacts identified in the *Status of Listed Resources*, 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 20$^{th}$ 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).

Climactic shifts also occur due to natural phenomena. In the North Atlantic, this primarily concerns fluctuations in the North Atlantic Oscillation, which results from changes in atmospheric pressure between a semi-permanent high pressure feature over the Azores and a subpolar low pressure area over Iceland (Curry and McCartney 2001; Hurrell 1995; Stenseth et al. 2002a). This interaction affects sea surface temperatures, wind patterns, and oceanic

circulation in the North Atlantic (Stenseth et al. 2002a). The North Atlantic Oscillation shifts between positive and negative phases, with a positive phase having persisted since 1970 (Hurrell 1995). North Atlantic conditions experienced during positive North Atlantic Oscillation phases include warmer than average winter weather in central and eastern North America and Europe and colder than average temperatures in Greenland and the Mediterranean Sea (Visbeck 2002). Effects are most pronounced during winter (Taylor et al. 1998). The North Atlantic Oscillation is significant for North Atlantic right whales due to its influence on the species primary prey, zooplankton of the genus *Calanus*, which are more abundant in the Gulf of Maine during positive North Atlantic Oscillation years (Conversi et al. 2001b; Greene and Pershing 2004; Greene et al. 2003a). This subsequently impacts the nutritional state of North Atlantic right whales and the rate at which sexually mature females can produce calves (Greene et al. 2003a).

### 7.2 Habitat degradation

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

Natural sources of ambient noise include: wind, waves, surf noise, precipitation, thunder, and biological noise from marine mammals, fishes, and crustaceans. Anthropogenic sources of ambient noise include: transportation and shipping traffic, dredging, construction activities, geophysical surveys, and sonars. In general, it has been asserted that ocean background noise levels have doubled every decade for the last six decades in some areas, primarily due to shipping traffic (IWC 2004). The acoustic noise that commercial traffic contributes to the marine environment is a concern for ESA-listed species because it may impair communication between individuals (Hatch et al. 2008), among other effects (Eriksen and Pakkenberg 2013; Francis and Barber 2013). For species inhabiting Arctic waters, vessel and industrial noise may become much more problematic as oil and gas development and commercial shipping lanes through ice-free areas expand and intensify (Reeves et al. 2014). Vessels pose not only a risk of ship strike, but also impede the ability of whales to communicate. Hatch et al. (2012) estimated that roughly two-thirds of a right whales' communication space may be lost due to current ocean noise levels, which have greatly increased due to shipping noise. Shipping noise is also linked with increased stress levels in right whales (Rolland et al. 2012b).

Marine debris is another significant concern for ESA-listed species and their habitats. Marine debris has been discovered to be accumulating in gyres throughout the oceans. Law et al. (2010) presented a time series of plastic content at the surface of the western North Atlantic Ocean and Caribbean Sea from 1986 to 2008. More than 60% of 6,136 surface plankton net tows collected small, buoyant plastic pieces. The data identified an accumulation zone east of Bermuda that is similar in size to the accumulation zone in the Pacific Ocean. Over half of cetacean species (including humpback, fin, sei, and sperm whales) are known to ingest marine debris (mostly plastic), with up to 31% of individuals in some populations containing marine debris in their guts and being the cause of death for up to 22% of individuals found stranded on shorelines (Baulch and Perry 2014).

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. 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). 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 1990c; O'Hara et al. 1988). This fundamentally reduces the reproductive potential of affected populations, many of which are already declining (such as loggerhead and leatherback sea turtle populations in the action area).

### 7.3 Dredging

Marine dredging vessels are common within U.S. coastal waters. Although the underwater noises from dredge vessels are typically continuous in duration (for periods of days or weeks at a time) and strongest at low frequencies, they are not believed to have any long-term effect on sea turtles. Hopper dredges in the dredging mode are capable of moving relatively quickly compared to sea turtle swimming speed and can thus overtake, entrain, and kill sea turtles as the suction draghead(s) of the advancing dredge overtakes the resting or swimming turtle. Entrained sea turtles rarely survive. Relocation trawling frequently occurs in association with dredging projects to reduce the potential for dredging to injure or kill sea turtles (Dickerson et al. 2007).

### 7.4 Seismic surveys

During October and November 2003, the NSF undertook a seismic survey over the mid-Atlantic Ridge. No marine mammals or sea turtles were observed during the cruise, which had airgun operations for six days (Holst 2004). The airgun array discharge size was 8,760 $in^3$.

There have also been numerous prior seismic surveys from 1979 to 2002. These include surveys with a 6-airgun, 1,350-$in^3$ array in 1990; a single, 45-$in^3$ GI gun in 1996 and 1998; and two 45-$in^3$ GI guns in 2002 (NSF 2014). Impacts to ESA-listed species were not identified.

A few weeks prior to the proposed seismic survey, another smaller seismic survey was being conducted by the *Langseth* along the New Jersey coast using an airgun array of 700-1,400 $in^3$. However, this survey ended after roughly 20 hours of active airgun use due to mechanical issues. No data are yet available as to impacts of the survey on ESA-listed resources.

Even with the likelihood of previous exposures to seismic surveys, we have little information as to what response individuals would have to future exposures to seismic sources. Based upon the little information available to us for marine mammals, if prior exposure produces a learned response, then this response would likely be similar to or less than prior responses to other stressors where the individual experienced a stress response associated with the novel stimuli and responded behaviorally as a consequence (such as moving away and reduced time budget for activities otherwise undertaken) (Andre and Jurado 1997; André et al. 1997; Gordon et al. 2006). We do not believe sensitization would occur based upon the lack of severe responses previously observed in marine mammals and sea turtles exposed to seismic sounds that would be expected to produce a more intense, frequent, and/or earlier response to subsequent exposures (see *Response Analysis*).

### 7.5 Vessel traffic

Vessel noise could affect marine animals in the study area. Shipping and seismic noise generally dominates ambient noise at frequencies from 20 to 300 Hz (Andrew et al. 2002; Hildebrand 2009; Richardson et al. 1995c). Background noise has increased significantly in the past 50 years as a result of increasing vessel traffic, and particularly shipping, with increases of as much as 12

dB in low frequency ranges; background noise may be 20 dB higher now versus preindustrial periods (Hildebrand 2009; Jasny et al. 2005; McDonald et al. 2006; NRC 1994; NRC 2003; NRC 2005; Richardson et al. 1995a). Over the past 50 years, the number of commercial vessels has tripled, carrying an estimated six times as much cargo (requiring larger, more powerful vessels)(Hildebrand 2009). Seismic signals emanating from sources a great distance from the action area also contribute to the low frequency ambient sound field (Hildebrand 2009). Baleen whales may be more sensitive to sound at those low frequencies than are toothed whales. Masking of acoustic information can result (Simard et al. 2013); an important issue for marine mammals that rely primarily on sound as a sense. Dunlop et al. (2010) found that humpback whales shifted from using vocal communication (which carries relatively large amounts of information) to surface-active communication (splashes; carry relatively little information) when low-frequency background noise increased due to increased sea state. Other coping mechanisms include shifting the frequency or amplitude of calls, increasing the redundancy or length of calls, or waiting for a quieter period in which to vocalize (Parks et al. 2013) (Boness et al. 2013; Holt et al. 2013). Increases in vessel traffic and marine industrial construction is associated with decreases in the presence of minke whales and gray seals, presumably due to increased noise in the area (Anderwald et al. 2013). Sonars and small vessels also contribute significantly to mid-frequency ranges (Hildebrand 2009).

### 7.6 U.S. Navy training and testing activities

The U.S. Navy conducts training and testing activities in multiple ranges along the U.S. east coast. A biological opinion completed in 2013 estimated the number of exposures of ESA-listed species to those activities that are expected to occur annually (Table 10).

**Table 10.** Anticipated incidental take of ESA-listed species within U.S. Navy East Coast Training Range Complexes (NMFS 2013).

| Whale or sea turtle species | Operating area | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | Northeast | | Virginia Capes | | Cherry Point | | Jacksonville | |
| | Harass | Harm | Harass | Harm | Harass | Harm | Harass | Harm |
| Blue | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Fin | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 |
| Humpback | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 |
| North Atlantic right | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Sei | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 |
| Sperm | 0 | 0 | 2 | 0 | 0 | 0 | 0 | 0 |
| Hardshell sea turtles | 0 | 0 | 300 | 2 | 0 | 0 | 11 | 1 |
| Kemp’s ridley | 0 | 0 | 555 | 5 | 0 | 0 | 2 | 0 |
| Leatherback | 0 | 0 | 9 | 0 | 0 | 0 | 11 | 1 |

| Whale or sea turtle species | Operating area | | | | | | | |
|---|---|---|---|---|---|---|---|---|
| | Northeast | | Virginia Capes | | Cherry Point | | Jacksonville | |
| | Harass | Harm | Harass | Harm | Harass | Harm | Harass | Harm |
| Northwest Atlantic loggerhead | 0 | 0 | 466 | 8 | 0 | 0 | 19 | 1 |

Anticipated impacts from these exposures include changes from foraging, resting, milling, and other behavioral states that require lower energy expenditures to traveling, avoidance, and behavioral states that require higher energy expenditures and, therefore, would represent significant disruptions of the normal behavioral patterns of the animals that have been exposed. Behavioral responses that result from stressors associated with these training activities are expected to be temporary and would not affect the reproduction, survival, or recovery of these species. Instances of harm identified generally represent animals that would have been exposed to underwater detonations at 205 dB re $\mu Pa^2$-s or 13 psi, which corresponds to an exposure in which 50% of exposed individuals would be expected to experience rupture of their tympanic membrane, an injury that correlates with measures of permanent hearing impairment (Ketten 1998c).

Training activities occurring within their Northeast, Virginia Capes, Cherry Point and Jacksonville Range Complexes that anticipated annual levels of take of ESA-listed species incidental to those training activities through 2014. U.S. Navy aerial bombing training in the ocean off the southeast U.S. involving live ordnance (500 and 1,000-lb bombs) has been estimated to have injured or killed 84 loggerhead, 12 leatherback, and 12 green or Kemp's ridley sea turtles (NMFS 1997). From 2009- 2012, NMFS issued a series of biological opinions to the U.S. Navy for training activities occurring within their Northeast, Virginia Capes, Cherry Point and Jacksonville Range Complexes that anticipated annual levels of take of ESA-listed species incidental to those training activities through 2014. During the proposed activities 2 fin whales, 2 humpback whales, 2 sperm whales, 344 hardshell sea turtles (any combination of green hawksbill, Kemp's ridley or Northwest Atlantic loggerhead sea turtles), 644 Kemp's ridley sea turtles, 21 leatherback sea turtles and 530 Northwestern Atlantic loggerhead sea turtles per year are expected to be harassed as a result of their behavioral responses to mid- and high frequency active sonar transmissions  Another six Kemp's ridley and five Northwestern Atlantic loggerhead turtles per year are expected to be injured during exposure to underwater detonations.

### 7.7 U.S. Marine Corps training in the Cherry Point Range Complex

Table 11 identifies the likely take associated with Marine Corps activities in the Cherry Point Range Complex.

**Table 11.** Incidental take associated with U.S. Marine Corps training in the Cherry Point Range Complex that is currently authorized.

| Species | MCAS Cherry Point water ranges | | | | | | |
|---|---|---|---|---|---|---|---|
| | Boat maneuvers (BT-9 & BT-11) | | Ordnance/munitions delivery (BT-9 & BT-11) | | Underwater explosions (BT-9 only) | | |
| | Harass | Harm (injury, mortality) from vessel strike | Harass | Harm (injury, mortality) from direct strike | Harass (TTS and other behavioral impacts) | Harm | |
| | | | | | | Injury | Mortality |
| Green sea turtle<br>Kemp's ridley sea turtle<br>Leatherback sea turtle<br>Northwest Atlantic DPS Loggerhead sea turtle | 10 of any species per year | 1 of any species over a 10-year period | 10 of any species per year | 2 of any species over a 10-year period | 23 per year | 1 per year (PTS) | 1 over a 10-year period |

## 7.8 Entrapment and entanglement in fishing gear

Fisheries interactions are a significant problem for several marine mammal species and particularly so for humpback whales, as well as sea turtles. Between 1970 and 2009, two-thirds of mortalities of large whales in the northwestern Atlantic were attributed to human causes, primarily ship strike and entanglement (Van der Hoop et al. 2013). In excess of 97% of entanglement is caused by derelict fishing gear (Baulch and Perry 2014). Aside from the potential of entrapment and entanglement, there is also concern that many marine mammals that die from entanglement in commercial fishing gear tend to sink rather than strand ashore, thus making it difficult to accurately determine the frequency of mortalities. Entanglement may also make whales more vulnerable to additional dangers, such as predation and ship strikes, by restricting agility and swimming speed. Like fin whales, humpback whales have been entangled by fishing gear off Newfoundland and Labrador, Canada. A total of 595 humpback whales were reported captured in coastal fisheries in those two provinces between 1969 and 1990, of which 94 died (Lien 1994; Perkins and Beamish 1979). Along the Atlantic coast of the U.S. and the Maritime Provinces of Canada, there were 160 reports of humpback whales being entangled in fishing gear between 1999 and 2005 (Cole et al. 2005c; Nelson et al. 2007c). Of these, 95 entangled humpback whales were confirmed, with 11 whales sustaining injuries and nine dying of their wounds. Waring et al. (2007) reported four fin whales in the western North Atlantic having died or were seriously injured in fishing gear

Of the current threats to North Atlantic right whales, entanglement in commercial fishing gear poses one of the greatest threats (Figure 7). Along the Atlantic coast of the U.S. and the Maritime Provinces of Canada, there were 46 confirmed reports of North Atlantic right whales entangled in fishing gear between 1990 and 2007 (Cole et al. 2005a; Nelson et al. 2007b; Waring et al.

2009). Of the 39 reports that NMFS could confirm, North Atlantic right whales were injured in five of the entanglements and killed in four entanglements. Three of the 24 entangled whales between 2004 and 2008 died and one other resulted in serious injury (Glass et al. 2009). Recent efforts to disentangle right whales have met with success (Anonmyous. 2009).

**Figure 7.** A North Atlantic right whale entangled in fisheries gear off Florida, with Georgia Department of Natural Resources and Coastwise Consulting staff attempting to cut rope off (Credit: EcoHealth Alliance and Georgia Department of Natural Resources, ESA permit number 932-1905).

Nine instances of entanglement were recorded between 2006 and 2010, two of which were disentangled (Waring et al. 2013). From 1970-2010, 74 instances of entanglement have been documented (Waring et al. 2013). Scars examined between 1980 and 2002 revealed that 75% of 447 individuals examined showed scarring from fishing gear (Waring et al. 2013). It is also estimated that 14 and 51% of right whales are entangled on an annual basis (Knowlton et al. 2005). Another study assessing photographs of right whales from 1980-2009 found 626 individuals having 1,032 entanglement scars (Knowlton et al. 2012). This included 83% having at least one scar and 59% having multiple scars, with juveniles being entangled at higher rates than adults and the sexes entangling equally (Knowlton et al. 2012). Scars also became more abundant over the study period, suggesting entanglement rates are increasing (Knowlton et al. 2012). In August 1993, a dead sperm whale, with longline gear wound tightly around the jaw, was found floating about 32 km off Maine.

Fishery interaction remains a major factor in sea turtle recovery and, frequently, the lack thereof.

Wallace et al. (2010) estimated that worldwide, 447,000 turtles are killed each year from bycatch in commercial fisheries. NMFS (2002a) estimated that 62,000 loggerhead sea turtles have been killed as a result of incidental capture and drowning in shrimp trawl gear. Although TEDs 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. The fisheries that have the most significant demographic effect on sea turtles are the Gulf of Mexico shrimp trawl fisheries. The estimated annual number of interactions and mortalities between sea turtles and shrimp trawls in the Gulf shrimp fisheries (state and federal) are believed to have declined versus prior regulations (Epperly et al. 2002; Nance et al. 2008) (Table 12). Although participants in this and other fisheries are required to use Turtle Exclusion Devices, which are estimated to reduce the number of sea turtles trawlers capture by as much as 97%, each year these fisheries are expected to capture about 185,000 sea turtles annually and kill about 5,000 of them. Loggerhead sea turtles account for most of these: capturing about 163,000 loggerhead sea turtles, killing almost 4,000 of them. However, more recent estimates suggest interactions and mortality has decreased from pre-regulatory periods, with a conservative estimate of 26,500 loggerheads captured annually in U.S. Atlantic fisheries causing mortality to 1,400 individuals per year (Finkbeiner et al. 2011). These are followed by green sea turtles: about 18,700 green sea turtles are expected to be captured each year with more than 500 of them dying as a result of their capture (NMFS 2002b). Each year, various fisheries capture about 2,000 loggerhead sea turtles in Pamlico Sound, of which almost 700 die (Finkbeiner et al. 2011). The action area and its surrounding region appears to be a location of moderate sea turtle longline bycatch relative to long-term global levels (Lewison et al. 2014).

**Table 12.** Estimated annual interactions between sea turtles and shrimp trawls in the Gulf of Mexico shrimp fisheries associated estimated mortalities based on 2007 Gulf effort data taken from Nance et al. (2008).

| Species | Estimated interactions | Estimated mortalities |
|---|---|---|
| Leatherback | 520 | 15 |
| Loggerhead | 23,336 | 647 |
| Kemp’s ridley | 98,184 | 2,716 |
| Green | 11,311 | 319 |

Mortality of leatherbacks in the U.S. shrimp fishery is now estimated at 54 turtles per year. Data collected by the Northeast Fisheries Science Center 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.

Portions of the Atlantic pelagic fisheries for swordfish, tuna, shark, and billfish also operate in the action area and capture and kill the second highest number of sea turtles along the Atlantic

coast. These fisheries include purse seine fisheries for tuna, harpoon fisheries for tuna and swordfish, commercial and recreational rod and reel fisheries, gillnet fisheries for shark, driftnet fisheries, pelagic longline fisheries, and bottom longline fisheries. Lewison et al. (2004) 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). Between 1986 and 1995, this fishery captured and killed one North Atlantic right whale, two humpback whales, and two sperm whales. Between 1992 and 1998, the longline components of these fisheries are estimated to have captured more than 10,000 sea turtles (4,585 leatherback sea turtles and 5,280 loggerhead sea turtles), killing 168 of these, disincluding sea turtles that might have died after being released (Johnson et al. 1999; Yeung 1999). Since then, all components of these fisheries are estimated to capture about 1,350 sea turtles each year, killing 345. Finkbeiner et al. (2011) estimated that annual bycatch interactions total 1,400 leatherbacks annually for U.S. Atlantic fisheries (resulting in roughly 40 mortalities).

On 4 July 2004, NMFS published a final rule to implement management measures to reduce bycatch and bycatch mortality of Atlantic sea turtles in the Atlantic pelagic longline fishery (6979 FR 40734). The management measures include mandatory circle hook and bait requirements and mandatory possession and use of sea turtle release equipment to reduce bycatch mortality. This is expected to have significantly reduced sea turtle mortality from pelagic longlines.

In 2008, the Southeast Fisheries Science Center observer programs and subsequent analyses indicated that the overall amount and extent of incidental take for sea turtles specified in the incidental take statement of the 2005 opinion on the reef fish fishery had been severely exceeded by the bottom longline component of the fishery (approximately 974 captures and at least 325 mortalities estimated for the period July 2006-2007). The Gulf of Mexico Fishery Management Council developed a long-term management strategy via a new amendment (Amendment 31 to the Reef Fish FMP). The amendment included a prohibition on the use of bottom longline gear in the Gulf of Mexico reef fish fishery, shoreward of a line approximating the 35-fathom contour east of Cape San Blas, Florida, from June through August; a reduction in the number of bottom longline vessels operating in the fishery via an endorsement program and a restriction on the total number of hooks that may be possessed onboard each Gulf of Mexico reef fish bottom longline vessel to 1,000, only 750 of which may be rigged for fishing. These changes are expected to greatly reduce the mortality of loggerhead sea turtles resulting from the operation of this fishery.

Observation of the directed highly migratory shark fisheries has been ongoing since 1994, but a mandatory program was not implemented until 2002. Neritic juvenile and adult loggerhead sea turtles are the primary species taken, but leatherback sea turtles have also been observed caught. From 1994-2002, observers covered 1.6% of all hooks, observing bycatch of 31 loggerhead, 4 leatherback, and 8 unidentified sea turtles with estimated annual average take levels of 30, 222, and 56, respectively (NMFS 2003).

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.

## 7.9 Wind energy

Efforts to develop wind energy facilities offshore of the U.S. east coast have increased over the past several years. The Bureau of Ocean Energy Management assumed that the entire area of each Mid-Atlantic Wind Energy Area would be leased based on the expressions of commercial wind energy interest received (BOEM 2012). Leases could be issued and site characterization and assessment activities started as early as 2012 (BOEM 2012). Site characterization and assessment activities would occur over a period of about 5.5 years per lease (BOEM 2012). The most advanced in development of these is the Cape Wind Energy project (Cape Cod, Massachusetts), which calls for 130 wind turbine generators. The Bureau of Ocean Energy Management approved a construction and operations plan for the project in 2011 (USDOI 2011). Another six-turbine system is proposed off New Jersey, for which state permits were issued in 2011 (Fisherman's Energy of New Jersey LLC 2011). Several leases have been issued that would allow for testing and investigation of wind resources at various sites (BOEM 2012). Significant ocean noise and vessel activity is associated with construction of facilities such as these, which numerous studies have shown to displace marine mammals from the area, but who generally return post-construction. It is not known whether migratory species deflect to avoid facilities such as these once constructed.

## 7.10 Entrainment in power plants

Sea turtles entering coastal or inshore areas have been affected by entrainment in the cooling-water systems of electrical generating plants. A comprehensive biological opinion that covers all power plant cooling water intakes was issued by the Services in May 2014, but does not identify the amount or extent of ESA-listed species expected to be taken. This evaluation will be undertaken on a case-by-case basis for each power plant.

## 7.11 Ship-strikes

Ship-strike is a significant concern for the recovery of ESA-listed whales and, to a lesser degree, sea turtles. Between 1970 and 2009, two-thirds of mortalities of large whales in the northwestern Atlantic were attributed to human causes, primarily ship strike and entanglement (Van der Hoop et al. 2013). Between 1999 and 2005, there were three reports of sei whales being struck by vessels along the U.S. Atlantic coast and Canada’s Maritime Provinces (Cole et al. 2005c; Nelson et al. 2007c). Two of these ship strikes were reported as having resulted in death. An update (unpublished data 1995–2011) ship strike inventory for the eastern seaboard indicates the following percentage of strikes by species: North Atlantic right whale (19%), humpback whale (28%), sei whale (6%), fin whale (17%), sperm whale (2%), and unknown species (16%). Based on the records available, large whales have been struck by ships off almost every coastal state in the U.S., although ship strikes are most common along the Atlantic Coast. More than half (56%) of the recorded ship strikes from 1975-2002 occurred off the coasts of the northeastern U.S. and Canada, while the mid-Atlantic and southeastern areas each accounted for 22% (Jensen and Silber 2003). According to Waring et al. (2007), five fin whales were killed or injured as a result of ship strikes between January 2000 and December 2004. Between 1999-2005, there were 15 reports of fin whales strikes by vessels along the U.S. and Canadian Atlantic coasts (Cole et al. 2005a; Nelson et al. 2007b). Of these, 13 were confirmed, resulting in the deaths of 11 individuals. Of 123 humpback whales that stranded along the Atlantic coast of the U.S. between 1975 and 1996, 10 (8.1%) showed evidence of collisions with ships (Laist et al. 2001).

In the Bay of Fundy, recommendations for slower vessel speeds to avoid right whale ship strike

appear to be largely ignored (Vanderlaan et al. 2008). However, new rules for seasonal (June through December) slowing of vessel traffic to 10 knots and changing shipping lanes by less than one nautical mile to avoid the greatest concentrations of right whales are expected to reduce the chance of humpback whales being hit by ships by 9%, fin whales by 42%, right whales by 62%, and sei whales by 17%; the same rule applies from November through April from Brunswick, Georgia to Jacksonville, Florida, where North Atlantic right whales go for calving and breeding. Speed rules also apply to medium and large ports along the eastern seaboard during this time frame when right whales migrate to and from northern feeding and southern breeding areas. Nearly a dozen shipping lanes transect through coastal waters of the southeastern U.S. from the North-South Carolina to Cape Canaveral, Florida. Modeling efforts suggest voluntary changes in "areas to be avoided" suggested by the International Maritime Organization will reduce right whale strikes over the Scotian Shelf from one lethal strike every 0.78-2.07 years to one every 41 years (Hoop et al. 2012). Part of the susceptibility of North Atlantic right whales to ship strike may be its propensity to remain just below the surface, invisible to vessels, but at significant risk to ship strike (Parks et al. 2011b).

We believe the vast majority of ship-strike mortalities go unnoticed, and that actual mortality is higher than currently documented; Kraus et al. (2005) estimated that 17% of ship strikes are actually detected. The magnitude of the risks commercial ship traffic pose to large whales in the proposed action areas has been difficult to quantify or estimate. We struggle to estimate the number of whales that are killed or seriously injured in ship strikes within the U.S. EEZ and have virtually no information on interactions between ships and commercial vessels outside of U.S. waters. With the information available, we know those interactions occur but we cannot estimate their significance to whale species.

Ship strikes are the largest single contributor to North Atlantic right whale deaths, accounting for approximately 35% of all known mortalities, even though right whales should be able to hear the sound produced by vessels (Ketten 1998a; Knowlton and Kraus 2001a; Laist et al. 2001; Richardson et al. 1995a). Some information suggests right whales respond only within very close proximity to ships (Nowacek et al. 2004a). Various types and sizes of vessels have been involved in ship strikes with large whales, including container/cargo ships/freighters, tankers, steamships, U.S. Coast Guard vessels, Navy vessels, cruise ships, ferries, recreational vessels, fishing vessels, whale-watching vessels, and other vessels (Jensen and Silber 2004b). Injury is generally caused by the rotating propeller blades, but blunt injury from direct impact with the hull also occurs. There have been 18 reports of North Atlantic right whales being struck by vessels between 1999 and 2005 (Cole et al. 2005b; Nelson et al. 2007a). Of the 17 reports that NMFS could confirm, right whales were injured in two of the ship strikes and killed in nine. Recent records show that from 2004-2008, there were 17 confirmed reports of North Atlantic right whales being struck with eight whales dying of their wounds and two additional right whales sustaining serious injuries (Glass et al. 2009). Deaths of females are especially deleterious to the ability of the North Atlantic right whale population to recover. For instance, in 2005, mortalities included six adult females, three of which were carrying near-term fetuses and four of which were just starting to bear calves, thereby representing a lost reproductive potential of as many as 21 individuals over the short term (Kraus et al. 2005). Between 1999 and 2006, ships are confirmed to have struck 22 North Atlantic right whales, killing 13 of these whales (Jensen and Silber 2003; Knowlton and Kraus 2001b; NMFS 2005c). From 1999 to 2003, an average of 2.6 right whales were killed per year from various types of anthropogenic factors, but mostly from

ship-strike (Waring et al. 2010). From 2000 to 2004, this increased to 2.8 annually and increased again from 2001 to 2005 to an average of 3.2 right whales (Waring et al. 2010). The most recent estimate of anthropogenic mortality and serious injury available showed a rate of 3.8 right whales per year from 2002 to 2006. Of these, 2.4 were attributed to ship strikes (Glass et al. 2008). Based on records collected between 1970 and 1999, about 60% of the right whales struck by ships along the Atlantic Coast of the U.S., 20% occurred in waters off the northeast states and 20% occurred in waters off the mid-Atlantic or southeast states (Knowlton and Kraus 2001b). Over the same time interval (1970 to 1999), these authors identified 25 (45%) unconfirmed serious injuries and mortalities from ship strikes. Of these, 16 were fatal interactions; two possibly fatal; and seven nonfatal. Based on these confirmed mortalities, ships are responsible for more than one-third (16 out of 45, or 36%) of all confirmed right whale mortalities (a confirmed mortality is one observed under specific conditions defined by NMFS). [4] Part of the susceptibility of this species to ship strike may be its propensity to remain just below the surface, invisible to vessels, but at significant risk to ship strike (Parks et al. 2011b).

Another study conducted over a similar period (1970 to 2002) examined 30 (18 adults and juveniles, and 12 calves) out of 54 reported right whale mortalities from Florida to Canada (Moore et al. 2005). Human interaction (ship strike or gear entanglement) was evident in 14 of the 18 adults examined, and trauma, presumably from vessel collision, was apparent in 10 out of the 14 cases. Trauma was also present in four of the 12 calves examined, although the cause of death was more difficult to determine in these cases. In 14 cases, the assumed cause of death was vessel collision; an additional four deaths were attributed to entanglement. In the remaining 12 cases, the cause of death was undetermined (Moore et al. 2005).

Sea turtle ship strikes are 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 and Gyuris 2006; Hazel et al. 2007; Work et al. 2010). This, combined with the massive level of vessel traffic in the Gulf of Mexico and coastal Atlantic, has the potential to result in frequent injury and mortality to sea turtles in the region (MMS 2007). Hazel et al. (2007) suggested that green sea turtles may use auditory cues to react to approaching vessels rather than visual cues, making them more susceptible to strike as vessel speed increases. Overall, ship strike is likely highly underestimated as a source of injury or mortality to sea turtles in the action area.

### 7.12 Commercial whaling

Large whale population numbers in the action areas were impacted by commercial exploitation historically, mainly in the form of whaling. Between 1969-1990, 14 fin whales were captured in

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4 There are four main criteria used to determine whether serious injury or mortality resulted from ship strikes: (1) propeller cut(s) or gashes that are more than approximately 8 cm in depth; (2) evidence of bone breakage determined to have occurred premortem; (3) evidence of hematoma or hemorrhaging; and (4) the appearance of poor health in the ship-struck animal

Knowlton, A. R., and S. D. Kraus. 2001b. Mortality and serious injury of northern right whales (Eubalaena glacialis) in the western North Atlantic Ocean. Journal of Cetacean Research and Management Special Issue 2:193-208..

coastal fisheries off Newfoundland and Labrador; of these seven are known to have died because of capture (Lien 1994; Perkins and Beamish 1979). Commercial whaling no longer occurs within the action area.

### 7.13 Scientific and research activities

Scientific research permits issued by the NMFS currently authorize studies of ESA-listed species in the North Atlantic Ocean, some of which extend into portions of the action area for the proposed project. Authorized research on ESA-listed whales includes close vessel and aerial approaches, biopsy sampling, tagging, ultrasound, and exposure to acoustic activities, and breath sampling. Authorized research on ESA-listed sea turtles includes capture, handling, and restraint, satellite, sonic, and passive integrated transponder (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 number of "takes" rarely approach the number authorized. Therefore, it is unlikely that the 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 whales and 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 13-22 describe the cumulative number of takes for each ESA-listed species in the action area authorized in scientific research permits.

**Table 13.** Blue whale takes in the North Atlantic.

| Year | Approach | Biopsy | Suction cup tagging | Implantable tagging | Exhalation sampling | Acoustic playback |
|---|---|---|---|---|---|---|
| 2009 | 655 | 25 | 90 | 45 | 0 | 2 |
| 2010 | 720 | 25 | 90 | 45 | 0 | 0 |
| 2011 | 620 | 25 | 90 | 45 | 0 | 0 |
| 2012 | 730 | 25 | 90 | 45 | 0 | 0 |
| 2013 | 6,300 | 630 | 1,255 | 540 | 80 | 0 |
| 2014 | 5,715 | 630 | 1,165 | 495 | 80 | 0 |
| 2015 | 5,715 | 630 | 1,165 | 495 | 80 | 0 |
| Total | 20,455 | 1,990 | 3,645 | 1,710 | 240 | 2 |

Permit numbers: 633-1778, 775-1875, 1036-1744, 1058-1733, 10014, 14451, 14856, 15575, 16109, 16239, 16325, 16388, and 17355.

**Table 14.** Fin whale takes in the North Atlantic.

| Year | Approach | Biopsy | Suction cup tagging | Implantable tagging | Exhalation sampling | Acoustic playback |
|---|---|---|---|---|---|---|
| 2009 | 1,671 | 170 | 75 | 0 | 0 | 2 |
| 2010 | 1,876 | 170 | 45 | 0 | 0 | 0 |
| 2011 | 1,776 | 170 | 45 | 0 | 0 | 0 |
| 2012 | 2,846 | 170 | 45 | 0 | 0 | 0 |
| 2013 | 9,551 | 1,215 | 1,315 | 495 | 340 | 0 |
| 2014 | 8,727 | 1,165 | 1,290 | 495 | 340 | 0 |
| 2015 | 8,727 | 1,165 | 1,290 | 495 | 340 | 0 |
| Total | 32,174 | 4,225 | 4,105 | 1,485 | 1,020 | 2 |

Permit numbers: 10014, 605-1904, 775-1875, 948-1692, 981-1707, 1036-1744, 1058-1733, 14
14451, 14586, 14856, 15575, 16109, 16239, 16325, 16388, 16473, and 17355.

**Table 15.** Humpback whale takes in the North Atlantic.

| Year | Approach | Biopsy | Suction cup tagging | Implantable tagging | Belt tag | Exhalation sampling | Acoustic playback |
|---|---|---|---|---|---|---|---|
| 2009 | 5,260 | 415 | 173 | 45 | 0 | 0 | 624 |
| 2010 | 5,568 | 415 | 173 | 45 | 0 | 0 | 600 |
| 2011 | 8,653 | 1,040 | 723 | 95 | 0 | 0 | 600 |
| 2012 | 8,419 | 1,040 | 723 | 95 | 125 | 0 | 600 |
| 2013 | 17,925 | 1,980 | 1,465 | 395 | 125 | 2,410 | 600 |
| 2014 | 16,800 | 1,880 | 1,440 | 395 | 125 | 2,410 | 600 |
| 2015 | 16,155 | 1,880 | 1,440 | 395 | 125 | 2,410 | 0 |
| Total | 78,780 | 8,650 | 6,137 | 1,465 | 500 | 7,230 | 3,624 |

Permit numbers: 605-1904, 633-1778, 775-1875, 948-1692, 981-1707, 1036-1744, 1058-1733, 1121-1900, 1128-1922, 10014, 13927, 14118, 14245, 14451, 14586, 14856, 15575, 15682, 16109, 16325, 16388, 16473, and 17355.

**Table 16.** Sei whale takes in the North Atlantic.

| Year | Approach | Biopsy | Suction cup tagging | Implantable tagging | Exhalation sampling | Acoustic playback |
|---|---|---|---|---|---|---|
| 2009 | 1,604 | 50 | 158 | 45 | 0 | 2 |
| 2010 | 1,604 | 50 | 158 | 45 | 0 | 0 |
| 2011 | 1,504 | 50 | 158 | 45 | 0 | 0 |
| 2012 | 1,664 | 50 | 158 | 45 | 0 | 0 |
| 2013 | 8,227 | 1,735 | 773 | 390 | 160 | 0 |
| 2014 | 6,933 | 1,735 | 640 | 345 | 160 | 0 |
| 2015 | 6,933 | 1,735 | 640 | 345 | 160 | 0 |
| Total | 28,469 | 5,405 | 2,685 | 1,260 | 480 | 2 |

Permit numbers: 605-1904, 633-1778, 775-1875, 1058-1733, 10014, 14118, 14451, 14856, 15575, 16109, 16239, 16325, 16388, 16473, and 17355.

**Table 17.** Sperm whale takes in the North Atlantic.

| Year | Approach | Biopsy | Suction cup tagging | Implantable tagging | Exhalation sampling | Acoustic playback |
|---|---|---|---|---|---|---|
| 2009 | 5,560 | 375 | 820 | 0 | 0 | 920 |
| 2010 | 4,110 | 400 | 520 | 0 | 0 | 120 |
| 2011 | 4,010 | 425 | 520 | 0 | 0 | 120 |
| 2012 | 1,950 | 125 | 10 | 0 | 0 | 0 |
| 2013 | 8,789 | 990 | 720 | 450 | 80 | 0 |
| 2014 | 7,789 | 890 | 710 | 450 | 80 | 0 |
| 2015 | 7,789 | 890 | 710 | 450 | 80 | 0 |
| Total | 32,086 | 4,095 | 4,010 | 1,350 | 240 | 1,160 |

Permit numbers: 633-1778, 775-1875, 909-1719, 948-1692, 981-1707, 1036-1744, 1121-1900, 10014, 14451, 14586, 14856, 15575, 16109, 16239, 16325, 16473, 17312, and 17355.

**Table 18.** Green sea turtle takes in the Atlantic Ocean.

| Year | Capture/handling /restraint | Satellite,sonic, or pit tagging | Blood/tissue collection | Lavage | Ultrasound | Captive experiment | Laparoscopy | Imaging | Mortality |
|---|---|---|---|---|---|---|---|---|---|
| 2009 | 3,093 | 3,093 | 3,009 | 1,860 | 555 | 66 | 74 | 72 | 6 |
| 2010 | 3,753 | 3,753 | 3,669 | 2,480 | 555 | 66 | 74 | 72 | 6 |
| 2011 | 4,255 | 4,255 | 3,505 | 2,990 | 564 | 66 | 74 | 72 | 20 |
| 2012 | 3,354 | 3,354 | 2,622 | 2,210 | 704 | 66 | 74 | 72 | 18.2 |
| 2013 | 5,001 | 5,001 | 4,325 | 3,654 | 1,903 | 91 | 398 | 396 | 4.2 |
| 2014 | 4,236 | 4,236 | 3,560 | 3,004 | 1,408 | 65 | 324 | 324 | 4.2 |
| 2015 | 4,210 | 4,210 | 3,540 | 3,004 | 1,408 | 65 | 324 | 324 | 4.2 |
| Total | 27,902 | 27,902 | 24,230 | 19,202 | 7,097 | 485 | 1,046 | 1,332 | 62.8 |

Permit numbers: 1450, 1462, 1501, 1506, 1507, 1518, 1522, 1526, 1527, 1540, 1544, 1551, 1552, 1570, 1571, 1576, 10014, 10022, 13306, 13307, 13543, 13544, 13573, 14506, 14508,14622, 14655, 14726, 14949, 15112, 15135, 15552, 15556, 15575, 15606, 15802, 16134, 16146, 16174, 16194, 16253, 16556, 16598, 16733, 17183, 17304, 17355, 17381, and 17506.

**Table 19.** Kemp's ridley sea turtle takes in the Atlantic Ocean.

| Year | Capture/handling /restraint | Satellite,sonic, or pit tagging | Blood/tissue collection | Lavage | Ultrasound | Captive experiment | Laparoscopy | Imaging | Mortality |
|---|---|---|---|---|---|---|---|---|---|
| 2009 | 1,394 | 1,394 | 1,195 | 425 | 371 | 56 | 53 | 53 | 5 |
| 2010 | 1,402 | 1,402 | 1,203 | 426 | 371 | 56 | 53 | 53 | 5 |
| 2011 | 2,210 | 2,210 | 1,368 | 976 | 400 | 56 | 53 | 53 | 9 |
| 2012 | 2,229 | 2,219 | 1,561 | 972 | 450 | 56 | 53 | 53 | 7.2 |
| 2013 | 2,836 | 2,852 | 2,190 | 1,627 | 990 | 116 | 213 | 218 | 3.2 |
| 2014 | 2,460 | 2,476 | 1,814 | 1,256 | 619 | 60 | 160 | 165 | 3.2 |
| 2015 | 2,283 | 2,299 | 1,669 | 1,256 | 619 | 60 | 160 | 165 | 3.2 |
| Total | 14,814 | 14,852 | 11,000 | 6,938 | 3,820 | 460 | 745 | 548 | 35.8 |

Permit numbers: 1462, 1501, 1506, 1507, 1526, 1527, 1540, 1544, 1551, 1552, 1570, 1571, 1576, 10014, 10022, 13306, 13543, 13544, 14508, 14726, 14506, 14622, 14655, 14726, 15112, 15135, 15552, 15566, 15575, 15606, 15802, 16134, 16194, 16253, 16556, 16598, 16733, 17183, 17304, 17355, 17381, and 17506.

**Table 20.** Leatherback sea turtle takes in the North Atlantic Ocean.

| Year | Capture/handling/restraint | Satellite, sonic, or pit tagging | Blood/tissue collection | Lavage | Ultrasound | Imaging | Laparoscopy | Mortality |
|---|---|---|---|---|---|---|---|---|
| 2009 | 1,357 | 1,357 | 1,331 | 197 | 188 | 0 | 0 | 2 |
| 2010 | 1,421 | 1,421 | 1,394 | 197 | 188 | 0 | 0 | 1 |
| 2011 | 1,709 | 1,709 | 1,682 | 197 | 189 | 0 | 0 | 3.4 |
| 2012 | 736 | 736 | 709 | 187 | 189 | 0 | 0 | 2.6 |
| 2013 | 842 | 835 | 808 | 312 | 254 | 65 | 65 | 1.6 |
| 2014 | 653 | 646 | 620 | 135 | 66 | 65 | 65 | 1.6 |
| 2015 | 647 | 640 | 620 | 135 | 66 | 65 | 65 | 1.6 |
| Total | 7,365 | 7,344 | 7,164 | 1,360 | 1,140 | 195 | 195 | 13.8 |

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, 16253, 16556, 16733, 17355, and 17506.

**Table 21.** 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 | 5,909 | 5,909 | 4,859 | 2,609 | 2,519 | 305 | 401 | 354 | 24.8 |
| 2014 | 4,762 | 4,762 | 3,712 | 1,495 | 1,543 | 105 | 292 | 240 | 24.8 |
| 2015 | 4,635 | 4,635 | 3,635 | 1,495 | 1,543 | 105 | 292 | 240 | 7.8 |
| Total | 38,188 | 38,188 | 32,134 | 10,759 | 11,345 | 1,315 | 1,512 | 1,301 | 431.4 |

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, 14622, 14655, 14726, 15112, 15552, 15566, 15575, 15606, 15802, 16134, 16146, 16194, 16253, 16556, 16598, 16733, 17183, 17304, 17355, 17381, and 17506.

**Table 22.** Hawksbill sea turtle takes in the Atlantic Ocean.

| Year | Capture/handling /restraint | Satellite,sonic, or pit tagging | Blood/tissue collection | Lavage | Ultrasound | Captive experiment | Laparoscopy | Imaging | Mortality |
|---|---|---|---|---|---|---|---|---|---|
| 2009 | 1,088 | 1,088 | 1,081 | 464 | 254 | 0 | 0 | 0 | 3 |
| 2010 | 1,424 | 1,424 | 1,417 | 534 | 254 | 0 | 0 | 0 | 3 |
| 2011 | 1,959 | 1,959 | 1,955 | 914 | 255 | 0 | 0 | 0 | 4.4 |
| 2012 | 1,462 | 1,456 | 1,452 | 904 | 255 | 0 | 0 | 0 | 3.6 |
| 2013 | 1,423 | 1,417 | 1,415 | 844 | 320 | 39 | 0 | 0 | 1.6 |
| 2014 | 1,114 | 1,108 | 1,106 | 550 | 66 | 39 | 0 | 0 | 1.6 |
| 2015 | 1,032 | 1,026 | 1,026 | 550 | 66 | 39 | 0 | 0 | 1.6 |
| Total | 9,502 | 9,484 | 9,452 | 4,760 | 1,470 | 117 | 0 | 0 | 18.8 |

Permit numbers: 1462, 1501, 1506, 1507, 1518, 1526, 1527, 1540, 1544, 1551, 1552, 1570, 1571, 1576, 1599, 10014, 10022, 13306, 13307, 13543, 13544, 14272, 14508, 14726, 14506, 14508, 14622, 14655, 14726, 14949, 15112, 15135, 15552, 15566, 15575, 15606, 15802, 16134, 16146, 16194, 16253, 16598, 16733, 17183, 17304, 17355, 17381, and 17506.

## 7.14 Physical and oceanographic features

The presence of key habitat features, such as shelter or foraging opportunities, are the primary reasons why ESA-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 ESA-listed species occur in the action area at the levels we observe or expect. This does not represent a stressor, but is instead an underlining principle for establishing why effects are what we expect them to be.

The continental shelf in the northern half of the region is a nearly uniform, smooth seafloor with an evenly-carved continental shelf edge (Backus 1987). The continental shelf slopes gently and is relatively shallow, marked by an abrupt increase in the seafloor gradient and ranges in water depth from 100 to 150 m. The average width of the continental slope from Georges Bank to Cape Hatteras is approximately 30 km but varies in size from 10 to 50 km (Tucholke 1987). The largest submarine canyon in the area is the Hudson Canyon, which is the most developed canyon on the U.S. Atlantic continental margin. Several other smaller canyon features occur just west of the region where the seismic surveys would take place. Submarine canyons are considered to be complex regions of the continental slope containing a much richer biodiversity compared to uniform, smooth seafloor areas. The abundance of nutrients introduced by the strong down flowing currents are factors leading to the biological richness found in canyons (Cooper et al. 1987).

The Blake Plateau is the largest physical feature of the southern half of the region, shaped by the largest oceanographic feature, the Gulf Stream. The continental margin off North Carolina extends over 300 km from shore (Newton et al. 1971). The continental shelf, known as the Florida-Hatteras Shelf south of Cape Hatteras, is narrow at its northern extent (about 45 km) but broadens steadily to about 105 km off Cape Fear (Newton et al. 1971). The shelf break off North Carolina ranges in depth from 55-180 m. The continental slope in the region is relatively smooth and splits in two on either side of the Blake Plateau. The eastern half of the slope merges with the Blake Escarpment while the western slope follows the coastline (Emery and Uchupi 1972; Tucholke 1987).

The Gulf Stream Current is a powerful surface current, carrying warm water into the cooler North Atlantic just south of the action area (Pickard and Emery 1990; Verity et al. 1993). Surface velocities range from 2-5 nautical miles per hour and the temperature is generally 25° to 28° C (Mann and Lazier 1991). Cape Hatteras is considered to be the dividing point between the oceanic provinces of the South Atlantic Bight and the Middle Atlantic Bight (Newton et al. 1971; Pickard and Emery 1990). The Gulf Stream is usually sharply defined on its west and north side but much less so on its east or south sides (Pickard and Emery 1990).

In general, the Gulf Stream flows parallel to shore from the Florida Straits to Cape Hatteras, where it flows northeastward past the Grand Banks away from land. While stratification of the water column and other factors may play a role, climactic factors such as the North Atlantic Oscillation likely cause variation in its position (Pershing et al. 2001; Schmeits and Dijkstra 2000). Wave-like meandering begins to occur at Cape Hatteras and increase as the current progresses offshore. North of Cape Hatteras, small gyres form that separate from the Gulf Stream as either warm- or cold-core rings (Mann and Lazier 1991). Between three and eleven warm-core rings are formed per year, each about 100 km across (García-Moliner and Yoder 1994), 1,000 m

in height (Mann and Lazier 1991), and lasting 11-399 days (García-Moliner and Yoder 1994; Pickard and Emery 1990). Warm-core rings bring warm water and associated plankton to colder inshore areas. Cold-core rings form when a cyclonic loop pinches off from the Gulf Stream, resulting in a counterclockwise rotating ring of cool slope water in the warm Sargasso Sea (Pickard and Emery 1990). Twice as many cold-core rings are formed as warm-core rings every year (Pickard and Emery 1990). They are larger (100-300 km across) and longer lasting (months to years) than warm-core rings (Pickard and Emery 1990). Frontal eddies commonly occur over the continental shelf, forming south of the action area and moving north and enclosing cold, nutrient rich upwelled water (Mann and Lazier 1991; Yoder et al. 1981). This leads to temporary, locally enhanced primary production that can support zooplankton and larger ESA-listed sea turtle and marine mammal foraging. The Gulf Stream region acts to facilitate transport of some species (through entrainment in its flow) and restrict it for others (bounding cold-water and warm-water species from moving further south or north, respectively)(Wishner et al. 1988b). In addition to the Gulf Stream, a longshore current moves south along the coast consisting of cold, less saline, but nutrient-rich water from the Chesapeake Bay (Dzwonkowski and Yan 2005; Gangopadhyay et al. 2005; Lentz et al. 2003; Marmorino et al. 2002; Shen et al. 2000).

Upwelling, which replaces warm, generally nutrient poor water with deeper, colder, relatively nutrient rich water, occurs frequently in association with the Gulf Stream moving over the Florida-Hatteras Shelf (Lee et al. 1991; Savidge 2004). During fall, winter, and spring in the North Atlantic Bight, upwelling is usually restricted to the outer shelf of the Gulf Stream, but in summer, upwelled water intrudes onto the continental shelf under the warmer, less dense shelf water, leading to upwelling and resultant increases in productivity (Atkinson and Yoder 1984; Lee et al. 1991).

A persistent front exists from the Mid-Atlantic Bight into New England waters due to the intersection of the continental shelf and slope. This surface manifestation of a thermohaline front extends year round from the surface downward, where it intersects the seafloor just shoreward of the shelf break (Halliwell Jr. and Mooers 1979). Phytoplankton production is enhanced at this frontal boundary, often with twice the concentration of phytoplankton found in adjacent waters (Ryan et al. 1999b).

An annual phenomenon in the Mid-Atlantic Bight is the formation of the “cold pool.” This mass of cooler water occurs over the continental shelf in summer and stretches from the Gulf of Maine to Cape Hatteras and is detectible from spring through fall (Linder et al. 2004). The cold pool usually exists near the seafloor between the 40 m and 100 m isobaths and extends up into the water column for about 35 m. Minimum temperatures for the cold pool occur in early spring and summer and range from 1.1º to 4.7º C.

The North Atlantic Oscillation affects sea surface temperatures, wind conditions, and ocean circulation throughout the North Atlantic Ocean (Stenseth et al. 2002b). The North Atlantic Oscillation is an intensity alteration of the atmospheric pressure between the semi-permanent high pressure center over the Azores Islands and the subpolar low-pressure center over Iceland (Curry and McCartney 2001; Stenseth et al. 2002b). Sea-level atmospheric pressure in the two regions tends to vary inversely, creating “positive” and “negative” phases. However, these phases are stable for years to decades. The North Atlantic Oscillation was generally positive from 1900 to 1950, mainly negative in the 1960s and 1970s, and mainly positive since 1970 (Hurrell et al. 2001).

The North Atlantic Oscillation also influences the latitude of the Gulf Stream Current and is largely responsible for its variable location. During positive North Atlantic Oscillation years, the Gulf Stream is farther east (Taylor and Stephens 1998). The flow rate of the Gulf Stream is also affected; during negative North Atlantic Oscillation years, the Gulf Stream System is not only shifted southward but weakened by up to 25-33% (Curry and McCartney 2001). The upper slope-water system off the U.S. east coast is affected by the North Atlantic Oscillation (Pershing et al. 2001). During low North Atlantic Oscillation periods, the Labrador Current intensifies, leading to the advance of cold slope water along the continental shelf as far south as the Mid-Atlantic Bight (Pershing et al. 2001). Intensity variability in another regionally important current, the Labrador Current, is linked to the effects of winter temperatures in Greenland and its surrounding waterways, sea-ice formation, and the relative balance between the formation of deep and intermediate water masses and surface currents. Although the North Atlantic Oscillation influences the northern North Atlantic most, its effects remain significant south through the Outer Banks (Hurrell et al. 2001).

The North Atlantic Oscillation strongly affects trophic groups in North Atlantic marine ecosystems (Drinkwater et al. 2003; Fromentin and Planque 1996). *Calanus* copepod temporal and spatial patterns are linked to the phases of the North Atlantic Oscillation (Fromentin and Planque 1996; Stenseth et al. 2002b); positive North Atlantic Oscillation indices are associated with increased *Calanus* copepod abundance in the Gulf of Maine and the corollary in negative North Atlantic Oscillation index years (Conversi et al. 2001a; Greene et al. 2003b). This has secondary effects, such as prey availability for North Atlantic right whales, which feed principally on *Calanus finmarchicus*. High *Calanus finmarchicus* abundance is linked to increased North Atlantic right whale calving rates (Greene et al. 2003b). Negative North Atlantic Oscillation indices are associated with abundances of cod, herring, and sardines: species that are important to other ESA-listed mysticetes (Drinkwater et al. 2003).

Primary productivity fluctuates little south of the Chesapeake Bay. Important nutrient sources include discharge from the Pamlico and Neuse Rivers (although movement into the marine environment is limited by Pamlico Sound) and the Chesapeake Bay (Lohrenz et al. 2003). Chlorophyll α concentrations decrease quickly away from the coast to less than 1 $mg/m^3$ beyond the shelf break in all seasons. However, transient upwelling events associated with intrusion of Gulf Stream waters onto the Florida-Hatteras Shelf can support phytoplankton increases (Flierl and Davis 1993; García-Moliner and Yoder 1994; Lohrenz et al. 1993).

While exact estimates of enhanced productivity vary with the life of each cold-core ring, primary production is approximately 50% greater in cold-core rings than in the Sargasso Sea (Mann and Lazier 1996). Warm-core rings vary in their physical, chemical, and biological composition over their lifetime, either by entrainment from surrounding water masses or in situ changes (García-Moliner and Yoder 1994). Entrainment of both warm water from the Gulf Stream and cold water from the shelf/slope causes an increase in primary production (García-Moliner and Yoder 1994).

Phytoplankton are single-celled organisms that form the base of marine food chains and whose occurrence and abundance are strongly driven by light, temperature, and nutrient conditions. As nutrients from river outflows near shore generally provide more nutrients than are present offshore, phytoplankton are generally more abundant nearshore. Although the North Atlantic is generally well mixed (nutrients are generally available), light levels tend to be low for phytoplankton, limiting their growth (Ryan et al. 1999a). However, spring time is a period with

reduced mixing and increasing light levels, meaning that phytoplankton tend to stay at the surface and are better able to photosynthesize, grow, and reproduce at exponential rates (Mann and Lazier 1991; Parsons et al. 1984; Ryan et al. 1999a). However, nutrients are eventually exhausted in surface waters by May and seasonal progression into winter returns the region to a light-limiting condition. During spring and summer, nectophytoplankton are dominant in northern areas but are replaced by nanophytoplankton during limiting conditions (Ryan et al. 1999b). Diatoms, cyanobacteria, cryptophytes, and prasinophytes make up most of the phytoplankton community in the southern portion of the region, although haptophytes and dinoflagellates are more common closer to shore (Lohrenz et al. 2003). Assemblages depend greatly on highly-variable currents (Lohrenz et al. 2003). Coccolithophores and pyrrhophyceans predominate in Gulf Stream waters, and are generally least abundant in winter. These organisms serve as prey for those animals that ESA-listed individuals prey upon, such as jellyfish, zooplankton, and schooling fishes.

Not only the water conditions, but intersections between water bodies (frontal boundaries) are important factors in biological productivity. This is the case year-round between the shelf and slope waters of the mid-Atlantic, but particularly during winter and spring (Ryan et al. 1999a; Ryan et al. 1999b).

Zooplankton, the next higher level in the marine food chain from phytoplankton and the prey of several ESA-listed whales and sea turtles, are generally higher in slope water versus other locations (Wiebe et al. 1987). Spring is a time of higher abundance temporally, particularly within the upper 200 m of the water column (Wiebe et al. 1987). However, zooplankton biomass abundance can increase when shelf water intrudes over slope water, creating a stratified water column. High nutrients and a shallow mixed layer set conditions for enhanced phytoplankton production, which subsequently aids zooplankton biomass increases. Copepods are the primary zooplankters dominate in New England shelf waters, and whose abundance is highest in spring on the outer shelf but highest in summer on the inner shelf (Flagg et al. 1984). *Calanus finmarchicus* and *Pseudocalanus* sp. are the predominant copepods over the outer shelf while the inner shelf has *Centropages typicus* and *Temora longicornis* predominating. The relatively large size of *Calanus* species and its annual cycle in New England waters makes it a major driver of New England marine ecosystem during spring (Flagg et al. 1984). Zooplankton concentrate in areas of increased primary productivity, such as along Gulf Stream frontal boundaries and eddy peripheries (Oschlies and Garcon 1998). Here, zooplankton abundance changes with seasons, phytoplankton abundance, and oceanographic conditions, but is generally higher in cold-core eddies and along fronts (Quattrini et al. 2005; Wormuth et al. 2000). When shelf water intrudes over slope water, high nutrient concentrations and a shallow mixed layer will give rise to enhanced primary production, which then fuels an increase in zooplankton biomass or secondary production.

# 8 EFFECTS OF THE PROPOSED ACTIONS

Pursuant to section 7(a)(2) of the ESA, federal agencies must insure, through consultation with NMFS, that their activities are not likely to jeopardize the continued existence of any ESA-listed species or result in the destruction or adverse modification of critical habitat. The proposed use of the *Langseth* and issuance of the incidental harassment authorization by the Permits and Conservation Division for “takes” of marine mammals during the seismic studies would expose

ESA-listed species to seismic airgun pulses, as well as sound emitted from a multi-beam bathymetric echosounder and sub-bottom profiler and other stressors. In this section, we describe the potential physical, chemical, or biotic stressors associated with the proposed actions, the probability of individuals of ESA-listed species being exposed to these stressors, and the probable responses of those individuals (given probable exposures) based on the best scientific and commercial evidence available. 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 ESA-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 ESA-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 as defined by the MMPA of ESA-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 MMPA 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 similar to the US Fish and Wildlife Service's regulatory definition of "harass"[5] 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 160 dB acoustic "take" definition for marine mammals 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 sources at levels that define "take" as long as it creates the probability of injury. In addition, individuals may respond in a variety of ways, some of which have more significant fitness consequences than others. For example, quick evasion of a seismic source would be more significant than slow travel away from the same stressor due to increased metabolic demands, stress responses, and potential for calf abandonment that this response could

5 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)

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.

## 8.1 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. acoustic interference from engine noise;
3. ship-strikes;
4. entanglement in towed hydrophone, magnetometer, expendable bathythermographs, or sonobuoy filament cable;
5. sound fields produced by airguns; and
6. sub-bottom profiler or multibeam echosounder.

### Stressors Determined to be Discountable or Insignificant

Based on a review of available information, we 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 *Langseth* or its smaller counterparts is unlikely to cause widespread, high dose contamination (excluding the remote possibility of severe damage to the vessel) that would impact ESA-listed species directly or pose hazards to their food sources.

The propulsion system of the *Langseth* is designed to be very quiet compared to other vessels to reduce interference with seismic activities. Although noise originating from vessel propulsion will propagate into the marine environment, this amount would be highly improbable. The *Langseth*'s passage past a whale or sea turtle would be brief and not likely to be significant in impacting any individual's ability to feed, reproduce, or avoid predators. Brief interruptions in communication via masking are possible, but unlikely given the habits of whales to move away from vessels, either as a result of engine noise, the physical presence of the vessel, or both (Lusseau 2006). The same can be said for the chase vessel.

The *Langseth* will be traveling at generally slow speeds (7.8-8.3 km/h), reducing the amount of noise produced by the propulsion system and the probability of a ship-strike (Kite-Powell et al. 2007; Vanderlaan and Taggart 2007). Our expectation of ship strike is sufficiently small to be discountable due to the hundreds of thousands of kilometers the *Langseth* has traveled without a ship strike, general expected movement of marine mammals away or parallel to the *Langseth*, as well as the generally slow movement of the *Langseth* during most of its travels (Hauser and Holst 2009; Holst 2009; Holst 2010; Holst and Smultea 2008a). We have concluded the potential for ship strike or acoustic interference from propulsion and machinery noise is highly improbable and is discountable. Therefore, these potential stressors are not likely to adversely affect ESA-listed species and are not considered further in this Opinion.

ESA-listed species could interact directly with the towed hydrophone streamers and these interactions have been documented. An example of an interaction with a seismic survey occurred during a 2011 survey in the eastern tropical Pacific. During this survey, a dead olive ridley sea turtle was recovered from the foil of towed seismic gear; it is unclear whether the sea turtle became lodged in the foil pre- or post mortem (Spring 2011). Observations of sea turtles investigating streamers and not becoming entangled is also available (Hauser et al. 2008; Holst and Smultea 2008a; Holst et al. 2005a; Holst et al. 2005b). Although the towed hydrophone streamers and magnetometer could come in direct contact with an ESA-listed species, entanglements are highly unlikely and considered highly improbable based upon investigation into the use of these devices during the activities of other oceanographic activities. Expendable bathythermographs and sonobuoy filament cable do not trail behind the *Langseth*, but drop roughly vertically into the water column. Although information exists to support the potential for marine animals to become entangled in vertical lines, the probability of any individual interacting with a line to a degree that a negative consequence results is very low given the duration lines associated with the proposed action will be in the water and the number of lines involved. Based on this, we find the risk of entanglement in towed hydrophone, magnetometer, expendable bathythermographs, or sonobuoy filament cable so low as to be discountable. Therefore, it is not likely to adversely affected ESA-listed species and will not be considered further in this Opinion.

#### Stressors Considered Further

This consultation focused on the following stressors from the proposed seismic activities not considered insignificant or discountable and may adversely affect ESA-listed species: (1) acoustic energy introduced into the marine environment by the airgun array; and (2) acoustic energy introduced by both the sub-bottom profiler and multibeam echosounder sonars.

### 8.2 Exposure Analysis

Exposure analyses identify the ESA-listed species that are likely to co-occur with the action area 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.

The Permits and Conservation Division applies acoustic thresholds to help determine at what point during exposure to seismic airguns (and other acoustic sources) marine mammals are "harassed," under the MMPA. For this consultation, we adopted the same thresholds to estimate the number of exposures ESA-listed marine mammals (i.e., blue, sei, fin, humpback, North Atlantic right, and sperm whales) that would be exposed to seismic airguns at a level that would be harassment under the ESA. These thresholds help to develop exclusion radii around a source and the necessary power-down or shut-down criteria. Our exposure analysis for green, hawksbill, leatherback, loggerhead, and Kemp's ridley sea turtles assumed that received levels greater than 166 dB re 1 $\mu Pa_{rms}$ would be significant enough to result in "take" pursuant to the ESA.

For all ESA-listed species, the USGS provided a rationale in its environmental assessment for their assumption that each exposure would generally be a unique animal rather than re-exposure of the same animal multiple times. This rationale is that there is a very limited potential of re-ensonifying the same location within the action area. We find that 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 several hundred to tens of kilometers away due to individual

or situational sensitivity or other rationale for why whales and sea turtles move (ex. feeding or breeding opportunities unrelated to effects of the proposed action). As such, it is reasonable to expect that some individuals will receive a single exposure and vacate the immediate area. This is particularly significant given that marine mammals and sea turtles tend to return to specific areas for foraging and breeding, or use particular migratory corridors. We expect that at least some individuals would return to the area once the seismic activity has ceased. Observations from previous seismic surveys support the likelihood that individuals will be re-exposed is very low, if at all. We expect the only occasions when re-exposure may occur is when individuals move away and happen to place themselves on another portion of the seismic survey trackline. However, this is unlikely to happen in other than random, rare cases and we expect the vast majority of animals would only be exposed once.

The USGS and NMFS's Permits and Conservation Division estimated the number of ESA-listed whales exposed to received levels ≥160 dB re 1 $\mu Pa_{rms}$. This method was based upon a product of animal density and ensonified area. The ESA Interagency Cooperation Division identified an additional data source and method to estimate the number of ESA-listed marine mammals and sea turtles that would be exposed to received levels that we would consider take (≥160 dB re 1 $\mu Pa_{rms}$ for marine mammals and 166 dB re 1 $\mu Pa_{rms}$ for sea turtles). We present each approach below, as well as their relative strengths, weaknesses, and resulting exposure estimates. Maximum radii associated with seismic airgun isopleth modeling were established at the maximum diving depth for listed species (2,000 m). As several species do not dive to this depth and, for those that do, we expect that individuals will rarely be found at this depth, the isopleth distance from the source array is likely to overestimate the exposure ESA-listed individuals are expected to experience.

The USGS and NMFS's Permits and Conservation Division used data from the Navy Operating Area density estimates detailed in DON (2007) for sperm, humpback (summer), and fin whales, which are based upon NMFS Northeast and Southeast regional sighting surveys from 1998-2007 conducted during the same seasons (spring and/or summer) as the proposed seismic survey. USGS imported a shapefile of the study area (outlined in yellow in Figure 5) into the Ocean Biogeographic Information System Spatial Ecological Analysis of Megavertebrate Populations (OBIS-SEAMAP) online database to estimate marine mammal densities in the action area. This database only produces density estimates within the U.S. EEZ and 80% of the 2014 seismic survey trackline and 90% of the 2015 seismic survey trackline occurs outside the U.S. EEZ. The USGS overlaid the seismic survey study area to overlap where OBIS-SEAMAP provided density estimates and calculated a mean density for this area for each marine mammal species expected to occur in the study area for spring and summer seasons, respectively. These densities were used to represent the expected density throughout the entirety of the action area, including that which extends outside the U.S. EEZ. Only fin, humpback (summer), and sperm whale densities were generated in this way. For blue, humpback (spring), North Atlantic right, and sei whales, the action agencies assumed that a single group would be exposed during each 2014 and 2015 phases of the seismic survey, respectively and used mean group size observed during surveys reported in the Cetacean and Turtle Assessment Program (1982a) and Atlantic Marine Assessment Program for Protected Species surveys from 2009-2013 as the expected number of exposures.

Strengths in OBIS-SEAMAP approach include:

- Substantially higher density resolution based exclusively upon data obtained from robustly designed biological surveys through the region conducted over extended periods (Best et al. 2012; Ropert-Coudert et al. 2010).
- Allows for calculation of mean and variance based upon a larger data sample size.
- Uses relatively robust habitat modeling in addition to the direct sighting data it incorporates.
- NMFS's Permit and Conservation Division was able to reproduce the density estimates provided by the USGS in their environmental assessment with the information USGS provided, although with a small degree of difference likely stemming from map and shape file geospatial projections.
- The modeling process produces sharp changes in density in some locations that are not expected to be based upon species occurrence, but rather are artifacts of habitat modeling components of OBIS-SEAMAP. However, these are not as apparent in the study area as in locations outside this region (these artifacts are much more apparent in the other approach we evaluated).

Weaknesses in the OBIS-SEAMAP approach include:

- Estimates are based within U.S. EEZ waters and are applied outside the U.S. EEZ, where a large majority of the seismic survey track line occurs, frequently hundreds of miles from the U.S. EEZ.
- Little survey effort is incorporated into OBIS-SEAMAP through much of where the southern portion of the study area enters the U.S. EEZ, which makes up a major component of data used to determine overall density (Ropert-Coudert et al. 2010).
- The USGS included the area within the U.S. EEZ that overlapped a broad "study area" in calculating density estimates. Although this increases the region considered in calculating density and reduces variance associated with small area sample size incorporating relatively high or low regions that can unnaturally skew overall estimates, it also incorporates area that is not necessarily a part of the action area.
- Finally, the "study area" boundaries themselves were defined by the USGS as a general area surrounding the planned seismic survey tracklines that would allow for a degree of flexibility in altering these tracklines as the project developed. Although the area included within the study area was highly influential in determining the density data that were included in USGS's analysis, defining the area to be included did not consider biological, oceanographic, or anticipated effects within it, likely resulting in a considerable degree of subjectivity in the resulting density estimates.

For blue, humpback (spring), North Atlantic right, and sei whales, OBIS-SEAMAP data were not available to estimate a density, or produced estimates of zero individuals exposed (Prieto et al. 2012). In these cases, the action agencies assumed that a single group would be exposed during each 2014 and 2015 phases of the seismic survey, respectively and used mean group size observed during surveys reported in the Cetacean and Turtle Assessment Program (1982a) and Atlantic Marine Assessment Program for Protected Species surveys from 2009-2013 as the

expected number of exposures. The Permits and Conservation Division provided the ESA Interagency Cooperation Division with a rationale for group size as an appropriate estimate of exposure in the action area. This included expectations of blue, sei, North Atlantic right, and humpback (spring) rarity in the action area, extrapolations of relative blue to fin whale density observed in other areas (roughly 1:10), and potential timing differences of the seismic survey legs in 2014 and 2015 resulting in the different likelihoods of these species being present in the action area in one year versus another.

The ESA Interagency Cooperation Division identified an additional density data source worth consideration. As part of its environmental compliance efforts, the U.S. Navy developed the Navy Marine Species Density Database (NMSDD). This database utilizes the same data incorporated into OBIS-SEAMAP, and additional habitat-based modeling datasets that provide density estimates that encompass the entire study and action area of the proposed seismic survey. We worked with the NMFS's Permits and Conservation Division during consultation to develop an analytical approach to determining density using NMSDD data.

Although the data themselves are not available for this consultation to allow for reproducing the outputs, these data and the NMSDD outputs of them have been evaluated and incorporated into U.S. Navy actions consulted on by the ESA Interagency Cooperation Division, where agreement with the U.S. Navy has allowed for close inspection and analysis. A technical report detailing the analytical process by which NMSDD density estimates were determined, as well as output maps of the densities themselves for the seismic survey action area are also available (http://aftteis.com/DocumentsandReferences/AFTTDocuments/SupportingTechnicalDocuments.aspx). The NMSDD database also models density for all ESA-listed whale species expected to occur in the action area, including those that were not available or conducted through OBIS-SEAMAP.

As the data themselves were not available for independent modeling, we used the maps generated for each species (available on a monthly, seasonal, or annual basis, depending upon species). We expect that the seismic survey trackline in 2014 would be followed during the NMSDD summer season therefore we used this season for estimating density. The 2015 trackline may be done anytime between the months of April and August. As the earlier portion of this timeframe corresponds to the spring season on NMSDD maps and the later corresponds to summer, we used the season that would yield the highest density estimate for each species, respectively, for density estimates in 2015. For NMSDD density maps estimated on a monthly rather than seasonal basis, we used the map that would produce the highest density of all possible months that the 2014 and 2015 trackline could be undertaken, respectively. Maps were downloaded and georeferenced in ArcGIS 10.2. We then imported shape files, provided by USGS, for the 160 dB re 1 $\mu Pa_{rms}$ isopleth around the planned seismic survey trackline for 2014 and 2015, respectively. This was overlaid onto the georeferenced NMSDD map for each species. For each year's area ensonified to at least 160 dB re 1 $\mu Pa_{rms}$ (and for each species), that area was divided into 20 equal segments along the length of the trackline (Figure 8). For each of the 20 segments, the highest density estimated to occur within that particular segment was determined and that density assigned to the entire segment. Once values were assigned to each of the 20 segments, a mean of these densities was calculated and assigned for that species in that year. The process was replicated for 2014 and 2015 for each ESA-listed marine mammal species.

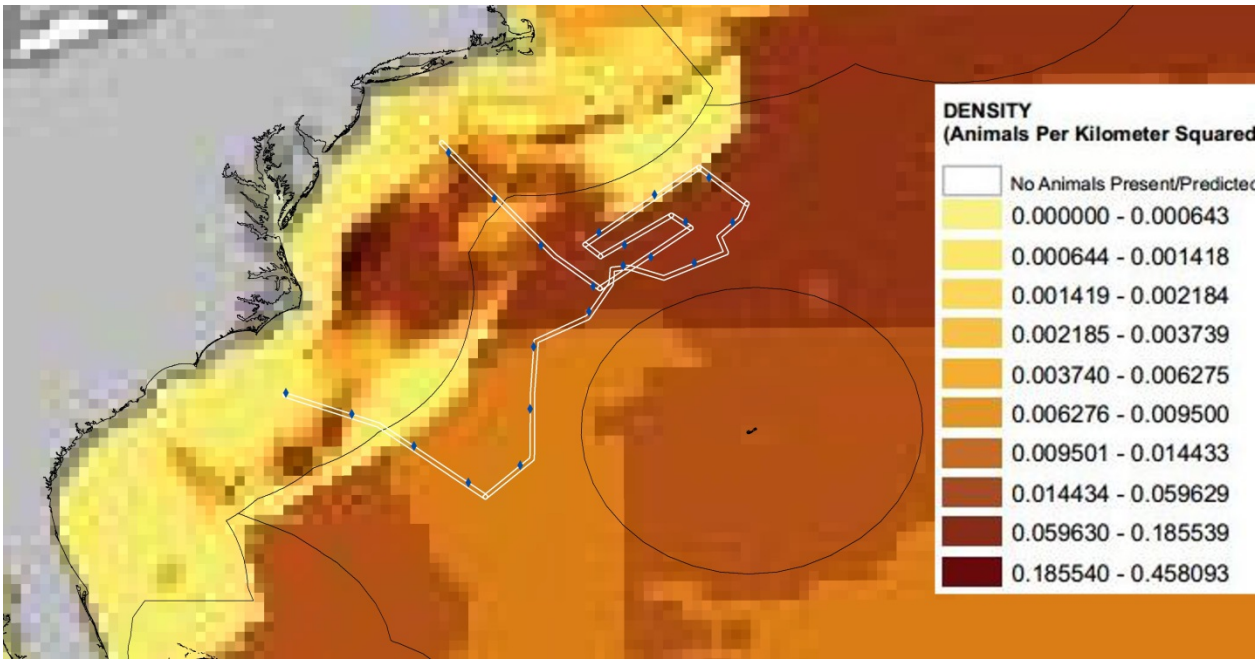


**Figure 8.** NMSDD summer sperm whale density estimate map georeferenced in ArcGIS 10.2 with area ensonified to at least 160 dB re 1 $\mu Pa_{rms}$ during 2014 seismic survey trackline. Blue diamonds demarcate start and end points between 20 equally-long segments of the 2014 survey trackline. Color shades (lightest to darkest) represent ranges of increasing density modeled within 10-40 $km^2$ squares from NMSDD. For the darkest shade identified in each of the 20 segments, the highest value of a particular density range was used as the value assigned to each of the 20 segments, respectively. Although the resolution of the map data are relatively course, we believe it provides sufficient detail to support the analytical process adopted in determining density in trackline segments.

Strengths in the NMSDD approach include:

- Greater density coverage of the action area.
- Density estimates are closer to the action area.
- All ESA-listed species of concern in this consultation are part of the database (marine mammals and sea turtles).
- More recent estimates of sightability and detectability of marine mammals.
- Although both datasets rely upon the same modeled data from within the U.S. EEZ, the NMSDD modeling extends density estimates through the entire action area.
- This dataset includes estimates derived from habitat usage information including continental slope waters and some abyssal areas driven more by continental shelf, slope, and nearshore physical and oceanographic processes.
- By assigning the highest value in a given range to a segment estimate, we do not risk underestimating the potential density and subsequent exposure or take given this density uncertainty.

Weaknesses in the NMSDD approach include:

- Underlying data are not available to us because data from outside the U.S. EEZ was licensed from a third party and not available from the U.S. Navy itself.
- The U.S. Navy itself expressed opinion that use of the NMSDD maps alone was not appropriate.
- The spatial resolution of the maps is gross (10-40 $km^2$ and likely somewhat more due to due to use of PDF maps) and could result in more subjectivity in the analysis.
- Density estimates outside the U.S. EEZ frequently show a sharp density gradient compared to values inside the U.S. EEZ. This is an artifact of the modeling process and is unlikely to reflect actual density.
- A degree of subjectivity is inherent in differentiating different color shades corresponding to density ranges on NMSDD maps, as shades can be difficult to distinguish at times.
- Map densities are represented as value ranges (generally two-to four fold difference between high and low values within a range) as opposed to pixel-based single value estimates, making estimates less accurate than OBIS-SEAMAP values in the U.S. EEZ.

We considered both approaches to estimate the number of ESA-listed animals that might be exposed to the seismic survey in this analysis.

The USGS estimated the exposure radii around the proposed *Langseth* operations using empirical data gathered in the Gulf of Mexico in 2007-2008 aboard the *Langseth*. The distances to which sound levels (rms) might propagate for single airgun and the 36-airgun array used during the proposed study are provided in Table 1. The maximum distance from airguns where received levels might reach 160 dB re 1 $\mu Pa_{rms}$ (36-airgun array) at 2,000 m depth (maximum depth at which ESA-listed species are expected to occur) is 5.78 km with a 9 m tow depth. A thorough review of available literature (see *Response Analysis*) supports this level as a general point at which baleen whales tend to show some avoidance response to received seismic sound.

The USGS's assumption that individuals will move away if they experience sound levels high enough to cause significant stress or functional impairment is also reasonable (see *Response Analysis*). Isopleth modeling tends to overestimate the distance to which various isopleths will propagate because most exposure will likely occur at depths shallower than 2,000 m, where received sound levels should be reduced (see Figures 2 and 3). Because we are unable to know where individuals will be in the water column at the time of exposure, we accept this assumption. In addition, the 160 dB re 1 $\mu Pa_{rms}$ radius will not always reach these distances, as shorter radii will occur during the use of smaller numbers of airguns (e.g., the use of a single airgun during power-down procedures). A received level of 166 dB re 1 $\mu Pa_{rms}$ (3.74 km for the 36-airgun array at nine meter tow depth) is considered here to be the threshold for harassment for sea turtle response based upon the little information available (McCauley et al. 2000a; McCauley et al. 2000b) (see sea turtle section below).

A major mitigation factor proposed by the USGS (and L-DEO) is visual monitoring, especially for marine mammals, which should reduce exposure of ESA-listed whales and sea turtles. However, visual monitoring has several limitations. Although regions ensonified by 160, 166, and 180 dB re 1 $\mu Pa_{rms}$ propagation distances are within the visual range of the *Langseth* and its

observers, it is unlikely that all ESA-listed species are easily visible at the surface at these distances. On their own, power-down and shut-down procedures are unlikely to be completely effective at eliminating the co-occurrence of listed individuals within the sound field ≥160 dB re 1 $\mu Pa_{rms}$ Other measures such as vessel turns and minimizing airgun source levels, seek to further minimize the exposure protected species will experience. Ramp-up was effective in reducing hearing-related effects in sonar systems (Von Benda-Beckmann et al. 2014) and we also expect reduced or less intense exposure in application to airgun ramp-up. When combined with the other proposed mitigation and monitoring measures, we conclude that the probability of listed individuals being exposed to the sound field ≥ 160dB re 1 $\mu Pa_{rms}$ is reduced significantly by the use of ramp-ups and shut-downs. Vessel platforms are subject to some limitations such as that even under good sighting conditions observers have limited ability to identify protected species during their brief time at the surface. Vocalizations by protected species will also help in identifying abundance of cetaceans in the action area. PAM will only detect the presence of marine mammals if they vocalize. Further ability to identify bearing, distance, and abundance is limited.

**Marine Mammals**

**Exposure of Listed Mammals to Airguns.** Exposure estimates from OBIS-SEAMAP and NMSDD whale density estimates and a planned ensonified area of approximately 36,587 $km^2$ in 2014 and 2015, respectively, along survey track lines, including areas of repeated exposure from adjacent track lines and turning legs.

The OBIS-SEAMAP exposure estimates (Tables 23 and 24) and NMSDD map exposure estimates (Tables 25 and 26) were calculated by using the density per 1,000 $km^2$ multiplied by the total survey track area (1,707 km track line ensonifying 36,587 $km^2$ to the 160 dB re 1 $\mu Pa_{rms}$ level in 2014 and 1,682 km track line ensonifying about the same area to the same level in 2015) to obtain the total number of exposures (rounded to the next whole number). For OBIS-SEAMAP exposure estimates, where density estimates were not available from the database or USGS qualitatively considered them to be low, group size for the species stemming from the Cetacean and Turtle Assessment Program (1982a) is substituted. As we cannot know whether 2015 activities will occur in spring or summer, the higher value of spring or summer is presented.

**Table 23.** 2014 estimated exposure of ESA-listed whales to sound levels ≥160 dB re 1 $\mu Pa_{rms}$ during the proposed seismic activities developed from OBIS-SEAMAP data and group size estimates.

| Whale density per 1,000 $km^2$ | # of exposures to listed whales | # of whales exposed to proposed activities | Population size | % of population exposed | Population/ location |
|---|---|---|---|---|---|
| **Blue-n/a** | 1 | Up to 1 | 440 | Up to 0.22% | Northwest Atlantic[1] |
| **Fin-0.06** | 3 | Up to 3 | 3,985 | Up to 0.08% | Northwest Atlantic[1] |

| Whale density per 1,000 km$^2$ | # of exposures to listed whales | # of whales exposed to proposed activities | Population size | % of population exposed | Population/ location |
|---|---|---|---|---|---|
| **Sei-n/a** | 3 | Up to 3 | 386 | Up to 0.78% | Nova Scotia stock[1] |
| **Humpback-n/a** | 3 | Up to 3 | 11,600 | Up to 0.03% | Northwestern Atlantic[2] |
| **North Atlantic right-n/a** | 3 | Up to 3 | 444 | Up to 0.68% | North Atlantic[1] |
| **Sperm-2.251** | 83 | Up to 83 | 13,190 | Up to 0.63% | Northeast Atlantic, Faroe Islands, Iceland, and northeastern U.S. coast[3] |
| **Total** | 131 | -- | -- | -- | -- |

[1] *Waring et al. (2013)*
[2] *IWC (2014)*
[3] *Whitehead (2002)*

**Table 24.** 2015 estimated exposure of ESA-listed whales to sound levels ≥160 dB re 1 μPa$_{rms}$ during the proposed seismic activities developed from OBIS-SEAMAP data and group size estimates.

| Whale density per 1,000 km$^2$ | # of exposures to listed whales | # of whales exposed to proposed activities | Population size | % of population exposed | Population/ location |
|---|---|---|---|---|---|
| **Blue-n/a** | 1 | Up to 1 | 440 | Up to 0.22% | Northwest Atlantic[1] |
| **Fin-0.06** | 3 | Up to 3 | 3,985 | Up to 0.08% | Northwest Atlantic[1] |
| **Sei-n/a** | 3 | Up to 3 | 386 | Up to 0.78% | Nova Scotia stock[1] |
| **Humpback-1.017** | 38 | Up to 38 | 11,600 | Up to 0.33% | Northwestern Atlantic[2] |
| **North Atlantic** | 3 | Up to 3 | 444 | Up to 0.68% | North Atlantic[1] |

| Whale density per 1,000 km$^2$ | # of exposures to listed whales | # of whales exposed to proposed activities | Population size | % of population exposed | Population/ location |
|---|---|---|---|---|---|
| **right-n/a** | | | | | |
| **Sperm-2.251** | 83 | Up to 83 | 13,190 | Up to 0.63% | Northeast Atlantic, Faroe Islands, Iceland, and northeastern U.S. coast[3] |
| **Total** | 131 | -- | -- | -- | -- |

[1] *Waring et al. (2013)*
[2] *IWC (2014)*
[3] *Whitehead (2002)*

**Table 25.** 2014 estimated exposure of ESA-listed whales to sound levels ≥160 dB re 1 μPa$_{rms}$ during the proposed seismic activities developed from NMSDD maps.

| Whale density per 1,000 km$^2$ | # of exposures to listed whales | # of whales exposed to proposed activities | Population size | % of population exposed | Population/ location |
|---|---|---|---|---|---|
| **Blue-0.035** | 2 | Up to 2 | 440 | Up to 0.45% | Northwest Atlantic[1] |
| **Fin-0.603** | 23 | Up to 23 | 3,985 | Up to 0.58% | Northwest Atlantic[1] |
| **Sei-5.851** | 215 | Up to 215 | 386 | Up to 55.70% | Nova Scotia stock[1] |
| **Humpback-2.104** | 77 | Up to 77 | 11,600 | Up to 0.66% | Northwestern Atlantic[2] |
| **North Atlantic right-0.023** | 1 | Up to 1 | 444 | Up to 0.22% | North Atlantic[1] |
| **Sperm-57.428** | 2,102 | Up to 2,102 | 13,190 | Up to 15.94% | Northeast Atlantic, Faroe Islands, Iceland, and northeastern U.S. coast[3] |

| Whale density per 1,000 km$^2$ | # of exposures to listed whales | # of whales exposed to proposed activities | Population size | % of population exposed | Population/ location |
|---|---|---|---|---|---|
| **Total** | 2,420 | -- | -- | -- | -- |

[1] *Waring et al. (2013)*
[2] *IWC (2014)*
[3] *Whitehead (2002)*

**Table 26.** 2015 estimated exposure of ESA-listed whales to sound levels ≥160 dB re 1 μPa$_{rms}$ during the proposed seismic activities developed from NMSDD maps.

| Whale density per 1,000 km$^2$ | # of exposures to listed whales | # of whales exposed to proposed activities | Population size | % of population exposed | Population/ location |
|---|---|---|---|---|---|
| **Blue-0.033** | 2 | Up to 2 | 440 | Up to 0.45% | Northwest Atlantic[1] |
| **Fin-5.762** | 211 | Up to 211 | 3,985 | Up to 5.29% | Northwest Atlantic[1] |
| **Sei-1.756** | 229 | Up to 229 | 386 | Up to 59.32% | Nova Scotia stock[1] |
| **Humpback-7.23** | 265 | Up to 265 | 11,600 | Up to 2.92% | Northwestern Atlantic[2] |
| **North Atlantic right-0.029** | 2 | Up to 2 | 444 | Up to 0.45% | North Atlantic[1] |
| **Sperm-57.692** | 2,111 | Up to 2,111 | 13,190 | Up to 16.00% | Northeast Atlantic, Faroe Islands, Iceland, and northeastern U.S. coast[3] |
| **Total** | 2,820 | -- | -- | -- | -- |

[1] *Waring et al. (2013)*
[2] *IWC (2014)*
[3] *Whitehead (2002)*

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 several hundred to tens of

kilometers away due to individual or situational sensitivity or other rationale for why whales move (ex. feeding or breeding opportunities unrelated to effects of the proposed action). As such, it is reasonable to expect that some individuals will receive a single exposure and vacate the action area by vacating the immediate area of the sound field. Other individuals may move, but move to locations where re-exposure could occur, either due to the direction or short distance they travel. Thus, it is reasonable that some individuals may be exposed more than once (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, and/or an individually randomly moves to a location that is later during the seismic survey). This is particularly significant given that marine mammals tend to return to specific areas for foraging and breeding, or use particular migratory corridors. However, based upon observations from previous seismic surveys and our professional judgment, the likelihood that individuals will be re-exposed several times is low. We expect the vast majority of exposed individuals will experience only a single exposure to the 160 dB re 1 $\mu Pa_{rms}$ level or higher.

Whales of all age classes are likely to be exposed. Based upon our understanding of listed whale life history presented in the *Status of Listed Resources*, listed whales are expected to be feeding, traveling, or migrating 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.

**Exposure of ESA-listed whales to multibeam echosounder and sub-bottom profiler.**

Two additional acoustic systems will operate during the proposed *Langseth* cruise: the multibeam echosounder and the sub-bottom profiler. These systems have the potential to expose ESA-listed species to sound above the 160 dB re 1 $\mu Pa_{rms}$ threshold. All systems operate at generally higher frequencies than airgun operations (10.5-13 kHz for the multibeam echosounder and 3.5 kHz for the sub-bottom profiler) and this mitigates effects since their frequencies will attenuate more rapidly than those from airgun sources. Listed individuals would experience higher levels of airgun noise well before either multibeam echosounder or sub-bottom profiler noise of equal amplitude would reach them. Thus, as explained below, operational airguns mitigate multibeam echosounder and sub-bottom profiler noise exposure.

The *Langseth* is expected to avoid close whale approaches, which reduces the chance of exposure to sonars as well. While airguns are not operational, marine mammal observers will remain on duty to collect sighting data. If ESA-listed whales were to approach the vessel, the *Langseth* would take evasive actions to avoid a ship-strike and simultaneously mitigate exposure to very high source levels. Ship strike has already been ruled out as an insignificant effect, and we also rule out high-level ensonification of listed whales (multibeam echosounder source level = 242 dB re 1 $\mu Pa_{rms}$; sub-bottom profiler source level = 204 dB re 1 $\mu Pa_{rms}$). Boebel et al. (2006) and Lurton and DeRuiter (2011) concluded that multibeam echosounders and sub-bottom profilers similar to those to be used during the proposed activities presented a low risk for auditory damage or any other injury, and that an individual would require exposure to 250–1,000 pulses from a sub-bottom profiler to be at risk for a temporary threshold shift (TTS). To be susceptible to TTS, a whale would have to pass at very close range and match the vessel's speed;

we expect a very small probability of this during the proposed study. An individual would have to be well within 100 m of the vessel to experience a single multibeam echosounder pulse that could result in TTS (LGL Ltd. 2008). The same result could only occur at even closer ranges for sub-bottom profiler signals, because the signals are weaker. Furthermore, we expect both multibeam echosounder and sub-bottom profiler systems to operate continuously with duty cycles of 1-20 s. It is possible, however, that some small number of listed whales (fewer than those exposed to airguns) could experience low-level multibeam echosounder and/or sub-bottom profiler sound. We are unable to quantify the level of exposure, but do not expect any exposure to occur at high levels.

**Sea Turtles**

**Exposure of ESA-listed turtles to airguns.** The USGS did not estimate the number or extent of exposure that would be expected for sea turtle species. We estimated exposure using the NMSDD density data maps previously described for whales and applied the same analytical process. However, we used the area ensonified to the 166 dB re 1 $\mu Pa_{rms}$ level instead of the 160 dB re 1 $\mu Pa_{rms}$ level. Based upon information presented in the *Response Analysis*, we expect all exposures at the 166 dB re 1 $\mu Pa_{rms}$ level and above to constitute "take" for sea turtles, not 160 dB re 1 $\mu Pa_{rms}$ as for whales. Also, NMSDD did not identify density for green or hawksbill sea turtles, as these species are difficult to differentiate at sea. NMSDD density estimates group green, hawksbill, and olive ridley (not expected to occur in the action area) sea turtles as "hardshell turtles" as a common estimate. We used the density value calculated for "hardshell sea turtles" to determine density for hawksbill and green sea turtles. We assigned a 6/7$^{ths}$ proportion of exposures to green sea turtles and 1/7$^{th}$ proportion to hawksbill sea turtles based upon the number of species-specific sightings (provided from OBIS-SEAMAP via the USGS environmental assessment) documented to occur in the action area during the same seasons as the proposed action. It is also important to note that NMSDD sea turtle density modeling does not extend as far offshore as it does for whales in NMSDD.

Our exposure estimates (Tables 26 and 27) were calculated by using the density per 1,000 km$^2$ multiplied by the total survey track area (1,707 km track line ensonifying 12,768 km$^2$ to the 166 dB re 1 $\mu Pa_{rms}$ level in 2014 and 1,682 km track line ensonifying about 12,581 km$^2$ to the same level in 2015) to obtain the total number of exposures (rounded to the next whole number). We also expect that the potential amount of re-exposure described for marine mammals applies to sea turtles.

We do not expect sound generated by the proposed action to expose eggs or hatchlings because we do not expect these life stages to be present in the action area. However, the *Status of Listed Resources* section identifies the oceanic environment of the North Atlantic as an important developmental habitat for juveniles and subadults of all sea turtle species and we expect these to occur in the action area. In addition, adult life stages of all species are expected to be exposed to sound. For sea turtle species that have been studied, a skewed sex distribution biased towards females versus males exists. As such, we expect more female sea turtles of all species to be exposed than males.

**Table 27.** 2014 estimated exposure of ESA-listed sea turtles to sound levels ≥166 dB re 1 $\mu Pa_{rms}$ during the proposed seismic activities.

| Sea turtle density per 1,000 km$^2$ | # of exposures to listed turtles | # of turtles exposed to proposed activities | Population size | % of population exposed | Population/ location |
|---|---|---|---|---|---|
| **Green-18.994** | 243 | Up to 243 | Unknown | Unknown | North Atlantic |
| **Hawksbill-3.166** | 41 | Up to 41 | Unknown | Unknown | North Atlantic |
| **Kemp's ridley-8.543** | 110 | Up to 110 | >189,000 | Up to 0.06% | North Atlantic[1] |
| **Leatherback–76.243** | 974 | Up to 974 | 34,000 | Up to 2.86% | North Atlantic[2] |
| **Loggerhead - 35.394** | 452 | Up to 452 | >32,000 | Up to 1.41% | Northwestern Atlantic[3] |
| **Total** | 1,820 | -- | -- | -- | -- |

[1]*Gallaway et al. (2013)*
[2]*TEWG (2007b)*
[3]*(NMFS 2001b; TEWG 1998c)*

**Table 28.** 2015 estimated exposure of ESA-listed sea turtles to sound levels ≥166 dB re 1 $\mu Pa_{rms}$ during the proposed seismic activities.

| Sea turtle density per 1,000 km$^2$ | # of exposures to listed turtles | # of turtles exposed to proposed activities | Population size | % of population exposed | Population/ location |
|---|---|---|---|---|---|
| **Green-66.205** | 833 | Up to 833 | Unknown | Unknown | North Atlantic |
| **Hawksbill-11.034** | 139 | Up to 139 | Unknown | Unknown | North Atlantic |
| **Kemp's ridley-12.407** | 102 | Up to 102 | >189,000 | Up to 0.05% | North Atlantic[1] |
| **Leatherback–41.958** | 528 | Up to 528 | 34,000 | Up to 1.55% | North Atlantic[2] |

| Sea turtle density per 1,000 km$^2$ | # of exposures to listed turtles | # of turtles exposed to proposed activities | Population size | % of population exposed | Population/ location |
|---|---|---|---|---|---|
| **Loggerhead - 101.357** | 1,276 | Up to 1,276 | >32,000 | Up to 3.99% | Northwestern Atlantic[3] |
| **Total** | 2,878 | -- | -- | -- | -- |

[1]*Gallaway et al. (2013)*
[2]*TEWG (2007b)*
[3]*(NMFS 2001b; TEWG 1998c)*

**Exposure of ESA-listed turtles to multibeam echosounder and sub-bottom profiler.** As with baleen whales, sea turtles hear in the low frequency range. The multibeam echosounder operates at 10.5-13 kHz and the sub-bottom profiler operates at 3.5 kHz, both of which are outside the hearing frequency of sea turtles. Thus, there is a low probability that sea turtles could experience exposure to sounds emitted by multibeam echosounder or sub-bottom profiler.

### 8.3 Response Analysis

As discussed in the *Approach to the Assessment* section of this Opinion, response analyses determine how ESA-listed resources are likely to respond after exposure to an action's effects on the environment or directly on ESA-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.

**Marine Mammals**

**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 ESA-listed whales and sea turtles considered in this Opinion. Possible responses considered in this analysis consist of:

- threshold shifts,
- auditory interference (masking),
- behavioral responses, and
- 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. A TTS results in a temporary hearing change and depends upon the duration, frequency, sound pressure, and rise time of the

sound (Finneran and Schlundt 2013). 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 ESA-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 $\mu Pa_{rms}$ in order to produce a low-level TTS from a single pulse (Southall et al. 2007b). PTS is expected at levels ~6 dB greater than TTS levels on a peak-pressure basis, or 15 dB greater on an SEL basis than TTS (Southall et al. 2007b). 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). If an individual experienced exposure to several airgun pulses of ~190 dB re 1 $\mu Pa_{rms}$, PTS could occur. A marine mammal would have to be within 100 m of the *Langseth's* airgun array to be within the 190 dB re 1 $\mu Pa_{rms}$ isopleth and risk a TTS. 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 $\mu Pa^2 s$.

Overall, we do not expect TTS or PTS to occur to any ESA-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 will 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 585 m 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 (Francis and Barber 2013). Masking can interfere with an individual's ability to gather acoustic information about its environment, such as predators, prey, conspecifics, and other environmental cues. Generally, noise will only mask a signal if it is sufficiently close to the signal in frequency. This can result in loss of environmental cues of predatory risk, mating opportunity, or foraging options (Francis and Barber 2013). 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 ESA-listed whales, particularly baleen whales. Any masking that might occur would likely be temporary because seismic sources are not continuous and the seismic vessel would continue to transit. The proposed seismic surveys could mask whale calls at some of the lower frequencies, in particular for baleen whales but also for sperm whales. This could affect communication between individuals, affect their ability to receive information from their environment, or affect sperm whale echolocation (Evans 1998; NMFS 2006h). Most of the energy of sperm whales clicks is concentrated at 2 to 4 kHz and 10 to 16 kHz, and though the findings by Madsen et al. (2006) suggest frequencies of seismic pulses can overlap this range, the strongest spectrum levels of airguns are below 200 Hz (0-188 Hz for the *Langseth* airguns). Given the disparity between sperm whale echolocation and communication-related sounds with the dominant frequencies for seismic surveys, masking is not likely to be significant for sperm whales (NMFS 2006h). Overlap of the dominant low frequencies of airgun pulses with low-frequency baleen whale calls would be expected to pose a greater risk of effects due to masking. The *Langseth's* airguns will emit a 0.1 s pulse when fired every 5 sec. 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 the *Marine mammals and behavioral responses* section below.

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 of responses and overall impact to be in the form of changes in behavior. 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 significant at the population level, but can equate to take. Displacement from important feeding or breeding areas over a prolonged period would likely be more significant. This has been suggested for humpback whales along the Brazilian coast as a result of increased seismic activity (Parente et al. 2007). Marine mammal responses to anthropogenic sound vary by species, state of maturity, prior exposure, current activity, reproductive state, time of day, and other factors (Ellison et al. 2012); this is reflected in a variety of aquatic, aerial, and terrestrial animal responses to anthropogenic noise that may ultimately have fitness consequences (Francis and Barber 2013). Although some studies are available which address responses of ESA-listed whales considered in this Opinion directly, additional studies to other related whales (such as bowhead and gray whales) are relevant in determining the responses expected by species under consideration. Therefore, studies from non-listed or species outside the action area are also considered here. Individual differences in responding to stressful stimuli also appear to exist and appear to have at least a partial genetic basis in trout (Laursen et al. 2011). Animals generally respond to anthropogenic perturbations as they would predators, increasing vigilance and altering habitat selection (Reep et al. 2011). Habitat abandonment due to anthropogenic noise exposure has been found in terrestrial species (Francis and Barber 2013).

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 operating locally (Greene Jr et al. 1999; Jochens et al. 2006; Madsen et al. 2002; McDonald et al. 1993; McDonald et al. 1995a; Nieukirk et al. 2004; Richardson et al. 1986; Smultea et al. 2004; Tyack et al. 2003). However, humpback whale males increasingly stopped vocal displays on Angolan breeding grounds as received seismic airgun levels increased (Cerchio et al. 2014). Some blue, fin, and sperm whales stopped calling for short and long periods apparently in response to airguns (Bowles et al. 1994; Clark and Gagnon 2006; McDonald et al. 1995a). Fin whales (presumably adult males) engaged in singing in the Mediterranean Sea moved out of the area of a seismic survey while airguns were operational as well as for at least a week thereafter (Castellote et al. 2012). A blue whale discontinued calls in response to received airgun sound of 143 dB re 1 μPa for one hour before resuming (McDonald et al. 1995a). Blue whales may also attempt to compensate for elevated ambient sound by calling more frequently during seismic surveys (Iorio and Clark 2009). Sperm whales, at least under some conditions, may be particularly sensitive to airgun sounds, as they have been documented to cease calling in association with airguns being fired hundreds of kilometers away (Bowles et al. 1994). Other studies have found no response by sperm whales to received airgun sound levels up to 146 dB re 1 $\mu Pa_{p\text{-}p}$ (Madsen et al. 2002; McCall Howard 1999). Some exposed individuals may cease calling in response to the *Langseth's* airguns. If individuals ceased calling in response to the *Langseth's* airguns during the course of the proposed survey, the effect would likely be temporary.

There are numerous studies of the responses of some baleen whale to airguns. Although responses to lower-amplitude sounds are known, most studies seem to support a threshold of ~160 dB re 1 $\mu Pa_{rms}$ as the received sound level to cause behavioral responses other than vocalization changes (Richardson et al. 1995c). Activity of individuals seems to influence response (Robertson et al. 2013), as feeding individuals respond less than mother/calf pairs and migrating individuals (Harris et al. 2007; Malme and Miles 1985; Malme et al. 1984; Miller et al. 1999; Miller et al. 2005; Richardson et al. 1995c; Richardson et al. 1999b). Surface duration decreased markedly during seismic sound exposure, especially while individuals were engaged in traveling or non-calf social interactions (Robertson et al. 2013). Migrating bowhead whales show strong avoidance reactions to received 120–130 dB re 1 $\mu Pa_{rms}$ exposures at distances of 20-30 km, but only changed dive and respiratory patterns while feeding and showed avoidance at higher received sound levels (152–178 dB re 1 $\mu Pa_{rms}$) (Harris et al. 2007; Ljungblad et al. 1988; Miller et al. 1999; Miller et al. 2005; Richardson et al. 1995c; Richardson et al. 1999b; Richardson et al. 1986). Responses such as stress may occur and the threshold for displacement may simply be higher while feeding. Bowhead calling rate was found to decrease during migration in the Beaufort Sea as well as temporary displacement from seismic sources (Nations et al. 2009). Bowheads were found to be less sightable during airgun exposure than at other times due to altered dive patterns (Robertson 2014). Calling rates decreased when exposed to seismic airguns at received levels of 116-129 dB re 1 μPa (possibly but not knowingly due to whale movement away from the airguns), but did not change at received levels of 99-108 dB re 1 μPa (Blackwell et al. 2013). Despite the above information and exposure to repeated seismic surveys, bowheads continue to return to summer feeding areas and when displaced, bowheads appear to reoccupy areas within a day (Richardson et al. 1986). We do not know whether the individuals exposed in these ensonified areas are the same returning or whether individuals that tolerate repeat exposures may still experience a stress response.

Gray whales respond similarly. Gray whales discontinued feeding and/or moved away at received sound levels of 163 dB re 1 $\mu Pa_{rms}$ (Bain and Williams 2006; Gailey et al. 2007; Johnson et al. 2007b; Malme and Miles 1985; Malme et al. 1984; Malme et al. 1986; Malme et al. 1988; Würsig et al. 1999; Yazvenko et al. 2007a; Yazvenko et al. 2007b). Migrating gray whales began to show changes in swimming patterns at ~160 dB re 1 μPa and slight behavioral changes at 140-160 dB re 1 $\mu Pa_{rms}$ (Malme and Miles 1985; Malme et al. 1984). As with bowheads, habitat continues to be used despite frequent seismic survey activity, but long-term effects have not been identified, if they are present at all (Malme et al. 1984). Johnson et al. (2007a) reported that gray whales exposed to seismic airguns off Sakhalin Island, Russia, did not experience any biologically significant or population level effects, based on subsequent research in the area from 2002–2005.

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

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

Sperm whale response to airguns has thus far included mild behavioral disturbance (temporarily 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. 2000d; Madsen et al. 2006; Miller et al. 2009; Moulton et al. 2006a; Moulton and Miller 2005; Stone 2003; Stone and Tasker 2006; Weir 2008). Detailed study of Gulf of Mexico sperm whales suggests some alteration in foraging from <130-162 dB re 1 $\mu Pa_{p-p}$, 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. Winsor and Mate (2013) did not find a nonrandom distribution of satellite-tagged sperm whales at and beyond five kilometers from seismic airgun arrays, suggesting individuals were not displaced or move away from the array at and beyond these distances in the Gulf of Mexico (Winsor and Mate 2013). However, no tagged whales within five kilometers were available to assess potential displacement within five kilometers (Winsor and Mate 2013). The lack of response by this species may in part be due to its higher range of hearing sensitivity and the low-frequency (generally <188 Hz) pulses produced by seismic airguns (Richardson et al. 1995c). Sperm whales are exposed to considerable energy above 500 Hz (Goold and Fish 1998). Breitzke et al. (2008) found that source levels were ~30 dB re 1 μPa lower at 1 kHz and 60 dB re 1 μPa lower at 80 kHz compared to dominant frequencies during a seismic source calibration. Another odontocete, bottlenose dolphins, progressively reduced their vocalizations as an airgun array came closer and got louder (Woude 2013). Reactions to impulse noise likely vary depending on the activity at time of exposure – e. g., in the presence of abundant food or during sexual encounters toothed whales sometimes are extremely tolerant of noise pulses (NMFS 2006b).

For whales exposed to seismic airguns during the proposed activities, behavioral changes stemming from airgun exposure may result in loss of feeding opportunities. We expect ESA-listed whales exposed to seismic airgun sound will exhibit an avoidance reaction, displacing individuals from the area at least temporarily. We also expect secondary foraging areas to be available that would allow whales to continue feeding. Although breeding may be occurring, we are unaware of any habitat features that sperm whales would be displaced from that is essential for breeding if sperm whales depart an area as a consequence of the *Langseth*'s presence. We expect breeding may be temporarily disrupted if avoidance or displacement occurs, but we do not expect the loss of any breeding opportunities. Individuals engaged in travel or migration would continue with these activities, although potentially with a deflection of a few kilometers from the route they would otherwise pursue.

**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 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; Gregory and Schmid 2001; Gulland et al. 1999; 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; Cattet et al. 2003; Dickens et al. 2010; Dierauf and Gulland 2001b; Elftman et al. 2007; Fonfara et al. 2007; Kaufman and Kaufman 1994; Mancia et al. 2008; Noda et al. 2007; Thomson and Geraci 1986). In some species, stress can also increase an individual's

susceptibility to gastrointestinal parasitism (Greer et al. 2005). 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 hypothalamic-pituitary-adrenal axis may persist for weeks (Dierauf and Gulland 2001a). Mammalian stress levels can vary by age, sex, season, and health status (Gardiner and Hall 1997; Hunt et al. 2006; Keay 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).

Loud noises generally increase stress indicators in mammals (Kight and Swaddle 2011b). 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. 2012a). These levels returned to baseline after 24 hours of traffic resuming. 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 ESA-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 2011b). Premature birth and indicators of developmental instability (possibly due to disruptions in calcium regulation) have been found in embryonic and neonatal rats exposed to loud sound. In fish eggs and embryos exposed to sound levels only 15 dB greater than background, increased mortality was found and surviving fry had slower growth rates (a similar effect was observed in shrimp), although the opposite trends have also been found in sea bream. Dogs exposed to loud music took longer to digest food. The small intestine of rats leaks additional cellular fluid during loud sound exposure, potentially exposing individuals to a higher risk of infection (reflected by increases in regional immune response in experimental animals). Exposure to 12 hours of loud noise can alter elements of cardiac tissue. In a variety of factors, including behavioral and physiological responses, females appear to be more sensitive or respond more strongly than males (Kight and Swaddle 2011b). 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 2011b).

**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. For more discussion regarding marine mammal strandings related to anthropogenic acoustic sources, please see (NMFS 2013).

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). We do not expect ESA-listed whales to strand as a result of the proposed seismic survey.

**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 ESA-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 fish and invertebrates 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). Nedelec et al. (Nedelec et al. 2014) found boat noise playbacks to cause significantly higher levels of mortality in early life stage sea hares. 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 on fishes and invertebrates. 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 μPa$^2$•s, but pike did show 10-15 dB of hearing loss with recovery within 1 day (Popper et al. 2005). Caged pink snapper have experienced PTS when exposed over 600 times to received seismic sound levels of 165-209 dB re 1 μPa$_{p\text{-}p}$. 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). Salmonid swim bladders (similar to the swim bladders of some marine mammal prey species) were reportedly damaged by received sound levels of ~230 dB re 1 μPa (Falk and Lawrence

1973).

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. Although received sound levels were not reported, caged *Pelates* spp., pink snapper, and trevally generally exhibited startle, displacement, and/or grouping responses upon exposure to airguns (McCauley and Fewtrell 2013a). This effect generally persisted for several minutes, although subsequent exposures to the same individuals did not necessarily elicit a response (McCauley and Fewtrell 2013a). 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 about two 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 one 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 $\mu Pa_{rms}$, 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). 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. (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 $\mu Pa_{rms}$ by first ejecting ink and then moving rapidly away from the area (McCauley and Fewtrell 2013b; 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 $\mu Pa_{rms}$. 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. Lobsters did not exhibit delayed mortality, or apparent damage to mechanobalancing systems after up to eight months post-exposure to airguns fired at 202 or 227 dB peak-to-peak pressure (Payne et al. 2013). However, feeding did increase in exposed individuals (Payne et al. 2013).

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 ESA-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 lethal or sub-lethal effects from airgun activities through reduced feeding opportunities for ESA-listed whales to be sufficient to reach a significant level. Effects are likely to be temporary and, if displaced, both sperm whales and their prey would re-distribute back into the area once survey activities have passed.

**Marine mammal response to multibeam echosounder and sub-bottom profiler.** We expect ESA-listed whales to experience ensonification from not only airguns, but also seafloor and ocean current mapping systems. Multibeam echosounder and sub-bottom profiler frequencies are much higher than frequencies used by all ESA-listed whales except blue, humpback, and sperm whales. We expect that these systems will produce harmonic components in a frequency range above and below the enter frequency similar to other commercial sonars (Deng et al. 2014). However, we do not expect these sub-harmonic frequencies in these systems to be audible to these species. Although Todd et al. (1992) found that mysticetes reacted to sonar sounds at 3.5 kHz within the 80-90 dB re 1 μPa range, it is difficult to determine the significance of this because the source was a signal designed to be alarming and the sound level was well below typical ambient noise. Goldbogen et al. (2013) found blue whales to respond to 3.5-4.0 kHz mid-frequency sonar at received levels below 90 dB re 1 μPa. Responses included cessation of foraging, increased swimming speed, and directed travel away from the source (Goldbogen et al. 2013). Hearing is poorly understood for ESA-listed baleen whales, but it is assumed that they are most sensitive to frequencies over which they vocalize, which are much lower than frequencies emitted by the multibeam echosounder and sub-bottom profiler systems (Ketten 1997; Richardson et al. 1995c). Thus, if fin, sei, or North Atlantic right whales are exposed, they are unlikely to hear these frequencies and a response is not expected.

Assumptions for blue, humpback, and sperm whale hearing are much different than for other ESA-listed whales. Humpback and sperm whales vocalize between 3.5-12.6 kHz and an audiogram of a juvenile sperm whale provides direct support for hearing over this entire range

(Au 2000a; Au et al. 2006; Carder and Ridgway 1990; Erbe 2002a; Frazer and Mercado 2000; Goold and Jones 1995; Levenson 1974; Payne and Payne 1985; Payne 1970; Richardson et al. 1995c; Silber 1986; Thompson et al. 1986; Tyack 1983; Tyack and Whitehead 1983; Weilgart and Whitehead 1993; Weilgart and Whitehead 1997; Weir et al. 2007; Winn et al. 1970). The response of a blue whale to 3.5 kHz sonar supports this species ability to hear this signal as well (Goldbogen et al. 2013). Maybaum (1990; 1993) observed that Hawaiian humpbacks moved away and/or increased swimming speed upon exposure to 3.1-3.6 kHz sonar. Kremser et al. (2005) concluded the probability of a cetacean swimming through the area of exposure when such sources emit a pulse is small, as the animal would have to pass at close range and be swimming at speeds similar to the vessel. Sperm whales have stopped vocalizing in response to 6-13 kHz pingers, but did not respond to 12 kHz echo-sounders (Backus and Schevill 1966; Watkins 1977; Watkins and Schevill 1975). Sperm whales exhibited a startle response to 10 kHz pulses upon exposure while resting and feeding, but now while traveling (Andre and Jurado 1997; André et al. 1997).

Investigations stemming from a recent stranding event in Madagascar suggest a 12 kHz multibeam echosounder, similar in operating characteristics as that proposed for use aboard the *Langseth*, suggest that this sonar played a significant role in the mass stranding of a large group of melon-headed whales (Southall et al. 2013). Although pathological data to suggest a direct physical affect are lacking and the authors acknowledge that although the use of this type of sonar is widespread and common place globally without noted incidents like the Madagascar stranding, all other possibilities were either ruled out or believed to be of much lower likelihood as a cause or contributor to stranding compared to the use of the multibeam echosounder (Southall et al. 2013). This incident highlights the caution needed when interpreting effects that may or may not stem from anthropogenic sound sources, such as the *Langseth*'s multibeam echosounder and that of the chase vessel. Although effects such as this have not been documented for ESA-listed species, the combination of exposure to this stressor with other factors, such as behavioral and reproductive state, oceanographic and bathymetric conditions, movement of the source, previous experience of individuals with the stressor, and other factors may combine to produce a response that is greater than would otherwise be anticipated or has been documented to date (Ellison et al. 2012; Francis and Barber 2013).

Recent stranding events associated with the operation of naval sonar suggest that mid-frequency sonar sounds may have the capacity to cause serious impacts to marine mammals. The sonars proposed for use by L-DEO differ from sonars used during naval operations, which generally have a longer pulse duration and more horizontal orientation than the more downward-directed multibeam echosounder and sub-bottom profiler. The sound energy received by any individuals exposed to the multibeam echosounder and sub-bottom profiler sources during the proposed activities is lower relative to naval sonars, as is the duration of exposure. The area of possible influence for the multibeam echosounder and sub-bottom profiler is also much smaller, consisting of a narrow zone close to and below the source vessel. Because of these differences, we do not expect these systems to contribute to a stranding event.

We do not expect masking of blue, sperm, or humpback whale communications to occur due to multibeam echosounder or sub-bottom profiler signal directionality, low duty cycle, and the brief period when an individual could be within its beam. These factors were considered when Burkhardt et al. (2013) estimated the risk of injury from multibeam echosounder was less than 3% that of ship strike.

## 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 one day.

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. 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.

**Sea turtles and behavioral responses.** As with ESA-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. 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 1994b; Lenhardt et al. 1983). However, loggerheads resting at the ocean surface were observed to 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 only expected when sound levels rise above received sound levels of 166 dB re 1 μPa. This corresponds with previous reports of sea turtle hearing thresholds being generally higher than for marine mammals (DFO 2004). At 166 dB re 1 μPa, we anticipate some change in swimming patterns and a stress response of exposed individuals. Some turtles may approach the active seismic array to closer proximity, but we expect them to eventually turn away. We expect temporary displacement of exposed individuals from some portions of the action area while the *Langseth* transects through.

**Sea turtles and stress.** Direct evidence of seismic sound causing stress is lacking in sea turtles. However, we expect sea turtles to generally 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. Therefore, we follow the best available evidence identifying a behavioral response as the point at which we also expect a significant stress response.

**Sea turtle response to multibeam echosounder and sub bottom profiler.** Sea turtles do not possess a hearing range that includes frequencies emitted by these systems. Therefore, ESA-listed sea turtles will not hear these sounds even if they are exposed and are not expected to respond to them.

# 9 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 ESA-listed resources into the foreseeable future. We expect climate change, habitat degradation, dredging, seismic surveys, military activities, entrapment and entanglement, invasive species impacts, wind energy projects, entrainment in power plants, ship-strikes, pollution, scientific research, and harvests to continue into the future. Movement towards bycatch reduction and greater foreign protections of sea turtles are generally occurring throughout the Atlantic Ocean, which may aid in abating the downward trajectory of sea turtle populations.

# 10 INTEGRATION AND SYNTHESIS OF EFFECTS

As explained in the *Approach to the Assessment* section, risks to ESA-listed individuals are measured using changes to an individual's "fitness" – i.e., the individual's growth, survival, annual reproductive success, as well as lifetime reproductive success. When ESA-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 ESA-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. Figure 4 provides a conceptual organization as to how we considered fitness consequences.

**ESA-listed whales.** The NSF proposes to allow the use of its vessel, the *Langseth*, to conduct a seismic survey by USGS that could incidentally harass several ESA-listed marine mammal species. These species include: blue whales, fin whales, humpback whales, North Atlantic right whales, sei whales, and sperm whales, all of whom are endangered throughout their ranges.

The *Status of Listed Resources* section identified commercial whaling as the primary reason for reduced populations, many of whom are a small fraction of their former abundance (Tables 3-7). Although large-scale commercial harvests no longer occur for these species, some harvests from subsistence and scientific research in regional and worldwide populations still occur. Other worldwide threats to the survival and recovery of ESA-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 North Atlantic face area-specific threats identified in the *Environmental Baseline*.

Despite these pressures, available trend information indicates most local populations of ESA-listed 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 ESA-listed whales by proposed seismic activities. We evaluated two approaches to estimate the number of ESA-listed animals that would be exposed to the seismic survey; each approach has advantages and disadvantages. Under the OBIS-SEAMAP and group size approach, we expect up to 2 blue, 6 fin, 6 sei, 41 humpback, 6 North Atlantic right, and 166 sperm whales could be exposed to airgun sounds during the course of the proposed seismic survey (2014 and 2015) which will elicit a behavioral response of temporarily moving out of the area. Under the approach using NMSDD maps, we expect up to 4 blue, 234 fin, 444 sei, 342 humpback, 3 North Atlantic right, and 4,213 sperm whales could be exposed to airgun sounds that would result in a similar response over the entirety of the seismic survey. We expect a low-level, transitory stress response to accompany this behavior. The number of individuals exposed based on the OBIS-SEAMAP and group size approach is expected to generally represent a small fraction of the populations. A larger proportion of sei and sperm whale populations would be expected to be exposed in the NMSDD map approach. We also consider that the population estimate (Nova Scotia stock) for sei whales is likely low, as the stock assessment includes only a small portion of the range that sei whales in the western Atlantic are expected to occur in, producing percent of population exposed estimates that are likely considerable overestimates.

The other actions we considered in the Opinion, the operation of multibeam echosounder and sub-bottom profiler systems, are not expected to be audible to fin, North Atlantic right, or sei whales and consequently are not expected to have any direct effects on these species. However, blue, humpback, and sperm whales could hear sounds produced by these systems. Responses could include cessation of vocalization by sperm whales and/or movement out of the survey area by these species.

Behavioral harassment caused by exposure to sound sources associated with the proposed seismic survey are expected to cause some individuals to cease these activities temporarily and

possibly move out of the immediate area. However, we expect that individuals will either resume foraging in a secondary location (which may be of somewhat lesser quality, but we cannot establish a defensible rationale for estimating it would be significantly so) or reoccupy the habitat from which they were displaced within a period of days. A metabolic cost associated with movement away from the sound sources may also occur, perhaps in most or all individuals exposed to 160 dB re 1 µPa levels or higher. However, as all ESA-listed marine mammal species in the action area routinely undertake long-distance movements in association with normal breeding and foraging patterns, we do not expect this to be meaningful to any individual's survival, growth, or reproductive potential.

These responses are expected from all individuals exposed and we do not expect a fitness consequence for any individual. Therefore, even though one exposure approach results in a much larger number estimates of exposure for some ESA-listed species, the proportion of population that experiences the response is not meaningful in determining jeopardy at the population or species level. Overall, we do not expect a fitness reduction to any individual whale. As such, we do not expect fitness consequences to populations or ESA-listed whale species as a whole.

**ESA-listed turtles.** Listed turtles that occur within the action area include green sea turtles, hawksbill sea turtles, leatherback sea turtles, loggerhead sea turtles, and Kemp's ridley sea turtles, which are either threatened or endangered. The *Status of Listed Resources* section found that most sea turtle populations have undergone significant to severe reduction by human harvesting of both eggs and turtles, as well as severe bycatch pressure in worldwide fishing industries. As previously mentioned, the *Cumulative Effects* section identified actions in the *Environmental Baseline* to generally continue for the foreseeable future.

From the *Effects Analysis*, we expect that 1,076 green, 180 hawksbill, 212 Kemp's ridley, 1,502 leatherback, and 1,728 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 all population affected. However, those that are available suggest a small proportion of each population would be affected. We expect that any response would be transient and of short duration and would not affect the fitness of any one individual. Therefore the proportions of the populations exposed are not relevant to determining jeopardy at the species level. We do not expect impairment of local nesting by the proposed survey. As we do not expect any sea turtle to be capable of hearing signals produced by the multibeam echosounder and sub-bottom profiler systems, we do not expect direct effects from these systems on sea turtle fitness. We do not anticipate any indirect effects from the proposed actions to influence sea turtles. Overall, we do not expect any individual sea turtle to undergo a fitness consequence.

# 11 CONCLUSION

After reviewing the current status of blue, fin, sei, humpback, North Atlantic right, and sperm whales as well as green, hawksbill, leatherback, loggerhead, and Kemp's ridley sea turtles; the *Environmental Baseline* for the action area; the anticipated effects of the proposed activities; and the *Cumulative Effects*, it is the NMFS's Opinion that USGS's proposed seismic survey using the NSF's vessel off the U.S. East Coast and NMFS's Permits and Conservation Division's issuance of an incidental harassment authorization pursuant to the MMPA for the seismic survey are not likely to jeopardize the continued existence of these species. The proposed action would have no

effect on critical habitat.

# 12 INCIDENTAL TAKE STATEMENT

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

The measures described below are nondiscretionary, and must be undertaken by the USGS and NMFS's Permits and Conservation Division so that they become binding conditions for USGS and L-DEO for the exemption in Section 7(o)(2) to apply. Section 7(b)(4) of the ESA requires that when a proposed agency action is found to be consistent with Section 7(a)(2) of the ESA and the proposed action may incidentally take individuals of ESA-listed species, NMFS will issue a statement that specifies the impact of any incidental taking of endangered or threatened species. To minimize such impacts, reasonable and prudent measures and terms and conditions to implement the measures, must be provided. Only incidental take resulting from the agency actions and any specified reasonable and prudent measures and terms and conditions identified in the incidental take statement are exempt from the taking prohibition of Section 9(a), pursuant to Section 7(o) of the ESA.

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's Permits and Conservation Division's proposed authorization of the incidental taking in the form of harassment of fin, blue, sei, humpback, North Atlantic right, and sperm whales pursuant to Section 101(a)(5)(D) of the MMPA. The final authorization will be issued and its mitigation and monitoring measures incorporated in this Incidental Take Statement as terms and conditions. With this authorization, the incidental take of ESA-listed whales is exempt from the taking prohibition of Section 9(a), pursuant to section 7(o) of the ESA as long as such take occurs consistent with this statement.

## 12.1 Amount or Extent of Take

The NMFS anticipates the proposed seismic survey along the U.S. East Coast is likely to result in the incidental take of ESA-listed species by harassment. In this case, we evaluated exposure and take through two approaches for marine mammals, both of which had advantages and disadvantages, and a single approach for sea turtles. These approaches produced markedly different take estimates. Under the OBIS-SEAMAP and group size approach, we expect up to 2 blue, 6 fin, 6 sei, 41 humpback, 6 North Atlantic right, and 166 sperm whales could be exposed to airgun sounds during the course of the proposed seismic survey (2014 and 2015) which will

elicit a behavioral response that would constitute harassment. Under an alternative analytical approach using NMSDD maps, we expect up to 4 blue, 234 fin, 444 sei, 342 humpback, 3 North Atlantic right, and 4,213 sperm whales could be exposed to airgun sounds that would result in a similar response over the entirety of the seismic survey. As detailed in the *Exposure Analysis*, all approaches have a large number of deficiencies that bring each into question as to being a best available approach. It was concluded that enough uncertainly exists that a single approach cannot be established as best. We also consider that our incidental take statement cannot exempt take where or to the extent that take is not authorized pursuant to the MMPA, as amended (16 U.S.C. 1371(a)(5)). As such, we cannot authorize more take for ESA-listed species than was included in the MMPA incidental harassment authorization.

We recognize that if we cannot identify a best approach, then we need to base our findings on what is the most conservative outcome for the ESA-listed resources in question. By authorizing the lowest estimated take numbers for each ESA-listed species as calculated from the various methods in front of us, it is reasonable to conclude that reinitiation will be triggered sooner than if larger estimates are adopted, thereby minimizing effects to ESA-listed species as a result of the proposed action. We acknowledge that this also means that future baseline conditions will not include the larger numbers associated with other approaches assessed here, which may impact the outcomes of future biological opinions. Based upon these considerations, the proposed action is expected to take by harassment 2 blue, 6 fin, 6 sei, 41 humpback, 3 North Atlantic right, and 166 sperm whales as well as 1,076 green, 180 hawksbill, 212 Kemp's ridley, 1,502 leatherback, and 1,728 loggerhead sea turtles by exposing individuals to received seismic sound levels greater than 160 dB re 1 μPa by harassment (166 dB re 1 μPa for sea turtles) utilizing the group size approach (blue and sei), OBIS-SEAMAP approach (fin, humpback and sperm whales), and NMSDD map approach (North Atlantic right whales and sea turtles) over the course of the project (2014 and 2015 combined). 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.

Harassment of blue, fin, humpback, North Atlantic right, sei, and sperm whales exposed to seismic studies at levels less than 160 dB re 1 μPa, or of leatherback, loggerhead, green, hawksbill, and Kemp's ridley sea turtles at levels less than 166 dB re 1 μPa, is not expected. If overt adverse reactions (for example, startle responses, dive reactions, or rapid departures from the area) by ESA-listed whales or sea turtles are observed at less intense levels than 160 dB or 166 dB re 1 μPa, respectively, while airguns are operating, incidental take may be exceeded. If such reactions by ESA-listed species are observed while airguns are in operation, this may constitute take that is not covered in this Incidental Take Statement. The USGS, L-DEO, NSF, and NMFS's Permits and Conservation Division must contact the ESA Interagency Cooperation Division to determine whether reinitiation of consultation is required because of such operations.

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

## 12.2 Effect of the Take

In the accompanying Opinion, NMFS has determined that the level of incidental take is not likely to jeopardize the continued existence of any ESA-listed species or result in the destruction or adverse modification of critical habitat**.**

## 12.3 Reasonable and Prudent Measures

NMFS believes the reasonable and prudent measure described below is necessary and appropriate to minimize the amount of incidental take of ESA-listed whales and sea turtles resulting from the proposed actions. This measure is non-discretionary and must be a binding condition of the USGS, L-DEO, and NMFS's authorization for the exemption in Section 7(o)(2) to apply. If the USGS, L-DEO, or NMFS fail to ensure compliance with this term and conditions and its implementing terms and conditions, the protective coverage of Section 7(o)(2) may lapse.

The USGS and L-DEO must implement and monitor the effectiveness of mitigation measures incorporated as part of the proposed authorization of the incidental taking of blue, fin, sei, humpback, North Atlantic right, and sperm whales pursuant to Section 101(a)(5)(D) of the MMPA and as specified below for green, hawksbill, Kemp's ridley, leatherback, and loggerhead sea turtles.

## 12.4 Terms and Conditions

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

To be exempt from the prohibitions of Section 9 of the ESA, the NSF, USGS, L-DEO, and Permits and Conservation Division must comply with the following terms and conditions, which implement the Reasonable and Prudent Measures described above and outlines the mitigation, monitoring and reporting measures required by the Section 7 regulations (50 CFR 402.14(i)). These terms and conditions are non-discretionary. If NSF, USGS, L-DEO, and/or the Permits and Conservation Division fail to ensure compliance with these terms and conditions and their implementing reasonable and prudent measures, the protective coverage of Section 7(o)(2) may lapse.

To implement the Reasonable and Prudent Measures, the USGS, L-DEO, and the NMFS's Permits and Conservation Division shall ensure that:

<u>Mitigation and Monitoring Requirements</u>

A. Establish an exclusion zone corresponding to the anticipated 180 dB re 1 $\mu Pa_{rms}$ isopleth for full (6,600 $in^3$) and single (40 $in^3$) airgun operations as well as a 160 dB re 1 $\mu Pa_{rms}$ buffer zone in which to document take and conduct mitigation.

B. Use two, NMFS-approved, vessel-based observers to watch for and monitor marine mammals and sea turtles near the seismic source vessel 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 and sea turtles, when practical. Observers will have access to reticle binoculars (7 X 50 Fujinon), big-eye binoculars (25 X 150), optical range finders, and night vision devices. Observer shifts will last no longer

than 4 hours at a time. Observers 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), Beaufort sea state and wind force, 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 exclusion zone using observers, for at least 30 min prior to starting the airgun (day or night). If observers find a marine mammal or sea turtle within the exclusion zone, USGS must delay the seismic survey until the marine mammal or sea turtle has left the area. If the observer sees a marine mammal or sea turtle that surfaces, then dives below the surface, the observer shall wait 30 minutes. If the observer sees no marine mammals or sea turtles during that time, they should assume that the animal has moved beyond the exclusion 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 or sea turtles are near, approaching or in the exclusion zone, the airguns may not be started up. If one airgun is already running at a source level of at least 180 dB re 1 $\mu Pa_{rms}$, L-DEO may start subsequent guns without observing the entire exclusion zone for 30 min prior, provided no marine mammals or sea turtles are known to be near the safety radius. While it is considered unlikely, in the event a North Atlantic right whale (*Eubalaena glacialis*) is visually sighted, the airgun array will be shut-down regardless of the distance of the animal(s) to the sound source. The array will not resume firing until 30 min after the last documented whale visual sighting. Concentrations (greater than or equal to three individuals that do not appear to be traveling) of humpback, sei, fin, blue, and/or sperm whales will be avoided if possible (*i.e.*, exposing concentrations of animals to 160 dB), and the array will be powered-down if necessary.

E. Use the PAM to detect marine mammals around the *Langseth* during all airgun operations and during most periods when airguns are not operating. One observer and/or bioacoustician will monitor the PAM at all times in shifts of 1-6 h. A bioacoustician shall design and set up the PAM system and be present to operate or oversee PAM, and available when technical issues occur during the survey.

F. Record the following when an animal is detected by the PAM:

i. Contact the observer immediately (and initiate power or shut-down, if required);

ii. Enter the information regarding the vocalization into a database. The data to be entered include an acoustic encounter identification number, whether it was linked with a visual sighting, date, time when first and last heard and whenever any additional

information was recorded, position and water depth when first detected, bearing if determinable, species or species group, types and nature of sounds heard (e. g., clicks, continuous, sporadic, whistles, creaks, burst pulses, strength of signal, etc.), and any other notable information.

G. Apply a "ramp-up" procedure when starting up at the beginning of seismic operations or any time after the entire array has been shut-down for more than 8 min, which means start the smallest gun first and add airguns in a sequence such that the source level of the array will increase in steps not exceeding approximately 6 dB per 5-min period. During ramp-up, the observers will monitor the 180 dB re 1 $\mu Pa_{rms}$ exclusion zone, and if marine mammals or sea turtles are sighted, a course/speed alteration, power-down, or shut-down will occur as though the full array were operational.

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

I. Shut-down or power-down the airguns upon marine mammal or sea turtle detection within, approaching, or entering the exclusion zone. A power-down means shutting down one or more airguns and reducing the buffer and exclusion zones to the degree that the animal is outside of one or both. Following a power-down, if the marine mammal or sea turtle approaches the smaller designated exclusion zone, the airguns must completely shut-down. Airgun activity will not resume until the marine mammal or sea turtle has cleared the exclusion zone, which means it was visually observed to have left the exclusion zone, or has not been seen within the exclusion zone for 15 min (small odontocetes) or 30 min (mysticetes, large odontocetes, and sea turtles). The *Langseth* may operate a small-volume airgun (*i.e.*, mitigation airgun) during turns and maintenance at approximately one shot per minute. During turns or brief transits between seismic tracklines, one mitigation airgun would continue to operate.

J. Marine seismic operations may continue into night and low-light hours if such segment(s) of the survey is initiated when the entire exclusion zone is 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 exclusion zone cannot be effectively monitored by the observer(s) on duty.

L. In the unanticipated event that the specified activity clearly causes any cases of marine mammal or sea turtle injury or mortality are judged to result from these activities (*e. g*., ship-strike, gear interaction, and/or entanglement), USGS will cease operating seismic airguns and report the incident to NMFS's Office of Protected Resources at 301-427-8401 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. If the lead observer judged that the injury or mortality is not a result of the authorized activities, operations may continue.

M. In the event that USGS discovers an injured or dead marine mammal or sea turtle, and the lead observer determines that the cause of the injury or death is unknown and the death is relatively recent (*i.e.*, in less than a moderate state of decomposition as described in the next paragraph), USGS will immediately report the incident to the Chief of the Permits and

Conservation Division, Office of Protected Resources, NMFS, at 301-427-8401, and/or by email to Jolie.Harrison@noaa.gov and Howard.Goldstein@noaa.gov, and (for marine mammals only) the NMFS Greater Atlantic Region Marine Mammal Stranding Network (866-755-6622) and/or by email to the NMFS Greater Atlantic Regional Stranding Coordinator (Mendy.Garron@noaa.gov), and the NMFS Southeast Region Marine Mammal Stranding Network (877-433-8299) and/or by e-mail to the Southeast Regional Stranding Coordinator (Blair.Mase@noaa.gov) and Southeast Regional Stranding Program Administrator (Erin.Fourgeres@noaa.gov). Activities may continue while NMFS reviews the circumstances of the incident. NMFS will work with USGS to determine whether modifications in the activities are appropriate.

N. In the event that USGS discovers an injured or dead marine mammal or sea turtle, and the lead PSO determines that the injury or death is not associated with or related to the activities authorized in Condition 2 of this Authorization (e. g., previously wounded animal, carcass with moderate to advanced decomposition, or scavenger damage), USGS shall report the incident to the Chief of the Permits and Conservation Division, Office of Protected Resources, NMFS, at 301-427-8401, and/or by email to Jolie.Harrison@noaa.gov and Howard.Goldstein@noaa.gov, and (for marine mammals only) the NMFS Greater Atlantic Marine Mammal Stranding Network (866-755-622), and/or by e-mail to the Greater Atlantic Regional Stranding Coordinator (Mendy.Garron@noaa.gov), and the NMFS Southeast Regional Stranding Network (877-433-8299), and/or by e-mail to the Southeast Stranding Coordinator (Blair.Mase@noaa.gov) and Southeast Regional Stranding Program Administrator (Erin.Fourgeres@noaa.gov), within 24 hours of the discovery. USGS shall provide photographs or video footage (if available) or other documentation of the stranded mammal sighting to NMFS and the Marine Mammal Stranding Network. Activities may continue while NMFS reviews the circumstances of the incident.

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

In addition, the proposed incidental harassment authorization requires L-DEO to adhere to the following reporting requirements:

B. 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 completion of 2014 activities and again after the completion of 2015 activities. The report would describe the proposed operations that were conducted and sightings of marine mammals and sea turtles within the vicinity of the operations. The report would provide full documentation of methods, results, and interpretation pertaining to all monitoring. The 90-day report would summarize the dates and locations of seismic operations, and all marine mammal and sea turtle sightings (i.e., dates, times, locations, activities, associated seismic survey activities, and associated PAM detections). This report must also contain and summarize the following information:

1. Summaries of monitoring effort – total hours, total distances, and distribution of marine mammals and sea turtles through the study period accounting for Beaufort sea state and wind force, and other factors affecting visibility and detectability of marine mammals and sea turtles;
2. Analyses of the effects of various factors influencing detectability of marine mammals and sea turtles including Beaufort sea state and wind force, number of observers, and

fog/glare;

3. Species composition, occurrence, and distribution of marine mammal and sea turtle sightings including date, water depth, numbers, age/size/gender, and group sizes; and analyses of the effects of seismic operations;
4. Sighting rates of marine mammal and sea turtles during periods with and without airgun activities (and other variables that could affect detectability);
5. Initial sighting distances versus airgun activity state;
6. Closest point of approach versus airgun activity state;
7. Observed behaviors and types of movements versus airgun activity state;
8. Numbers of sightings/individuals seen versus airgun activity state; and
9. Distribution around the source vessel versus airgun activity state.
10. The report would also include estimates of the number and nature of exposures that could result in "takes" of marine mammals and sea turtles by harassment or in other ways.

# 13 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 ESA-listed species or critical habitat, to help implement recovery plans, or to develop information.

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

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

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

# 14 REINITIATION NOTICE

This concludes formal consultation on the proposed seismic source survey to be carried out with the NSF's vessel and conducted by the USGS and L-DEO on board the *R/V Langseth* in the Atlantic Ocean off the U.S. East Coast, and the issuance of an incidental harassment authorization for the proposed studies pursuant to Section 101(a)(5)(D) of the MMPA. As provided in 50 CFR §402.16, reinitiation of consultation will be required where discretionary Federal involvement or control over the action has been retained or is authorized by law, and: (1)

if the amount or extent of incidental take is exceeded; (2) if new information reveals effects of the agency action that may affect ESA-listed species or critical habitat in a manner or to an extent not considered in this opinion; (3) if the agency action is subsequently modified in a manner that causes an effect to the ESA-listed species or critical habitat not considered in this opinion; or (4) if a new species is listed or critical habitat designated that may be affected by the action.

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