NATIONAL MARINE FISHERIES SERVICE
ENDANGERED SPECIES ACT SECTION 7 BIOLOGICAL OPINION

Action Agencies:	National Science Foundation-Division of Ocean Sciences and NOAA's National Marine Fisheries Service-Office of Protected Resources-Permits and Conservation Division
Activity Considered:	**1)** Seismic survey by the Lamont-Doherty Earth Observatory in the Eastern Mediterranean Sea, and **2)** Issuance of an Incidental Harassment Authorization pursuant to section 101(a)(5)(D) of the Marine Mammal Protection Act (MMPA)
Consultation Conducted By:	Endangered Species Act Interagency Cooperation Division, Office of Protected Resources, National Marine Fisheries Service
Approved:	Perry GAYALDO for Donna S. Wieting Director, Office of Protected Resources
Date:	OCT 16 2015
Public Consultation Tracking System (PCTS) number:	FPR-2015-9130

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ACRONYMS AND ABBREVIATIONS

AMAPPS-Atlantic Marine Assessment Program for Protected Species

BOEM-Bureau of Ocean Energy Management

CETAP-Cetacean and Turtle Assessment Program

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

EZ-Exclusion zone

HCB-Hexachlorobenzene

HMS FMP-Highly migratory species fisheries management plan

Hz-Hertz

IHA-Incidental harassment authorization

IPCC-Intergovernmental Panel on Climate Change

IWC-International Whaling Commission

kHz-kilohertz

kg-kilogram

L-DEO-Lamont Doherty Earth Observatory

MMPA-Marine Mammal Protection Act

ms-millisecond

NAO-North Atlantic oscillation

NMFS-National Marine Fisheries Service

NOAA-National Oceanic and Atmospheric Administration

NSF-National Science Foundation

PAM-Passive acoustic monitoring

PCB-Polychlorinated biphenyl

PFOA-Perfluorooctanoic acid

PFOS-Perfluorooctanesulfonic acid

PSVO-Protected species visual observer

PTS-Permanent threshold shift

RMS-Root mean squared

SEL-Sound exposure level

SERDP SDSS- Strategic Environmental Research and Development Program Spatial Decision Support System

TEWG-Turtle Expert Working Group

TTS-Temporary threshold shift

U.S.-United States

U.S.C.-United States Code

USFWS-United States Fish and Wildlife Service

1 INTRODUCTION

Section 7 (a)(2) of the ESA requires Federal agencies to insure that their actions are not likely to jeopardize the continued existence of endangered or threatened species or adversely modify or destroy their designated critical habitat. When a Federal agency's action "may affect" a protected species, that agency is required to consult formally with NOAA's National Marine Fisheries Service (NMFS) or the US Fish and Wildlife Service (USFWS), depending upon the endangered species, threatened species, or designated critical habitat that may be affected by the action (50 CFR §402.14(a)). Federal agencies are exempt from this general requirement if they have concluded that an action "may affect, but is not likely to adversely affect" endangered species, threatened species, or designated critical habitat and NMFS or the USFWS concurs with that conclusion (50 CFR §402.14(b)).

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

For the actions described in this document, the action agencies are the National Sciene Foundation (NSF) and the NMFS' Permits and Conservation Division. Two federal actions are considered in this biological opinion (Opinion). The first is the NSF's proposal 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 a seismic survey in the eastern Mediterranean sea near Greece from November to October of 2015, in support of an NSF-funded collaborative research project led by University of Oregon. The second is the NMFS' Permits and Conservation Division proposal to issue an incidental harassment authorization (IHA) authorizing non-lethal "takes" by Level B harassment (as defined by the Marine Mammal Protection Act (MMPA)) of marine mammals incidental to the planned seismic survey, pursuant to section 101 (a)(5)(D) of the MMPA, 16 U.S.C. § 1371 (a)(5)(D). The consulting agency is the NMFS' Office of Protected Resources, ESA Interagency Cooperation Division.

The opinion and incidental take statement were prepared by NMFS Endangered Species Act Interagency Cooperation Division in accordance with section 7(b) of the ESA and implementing regulations at 50 CFR § 402. This document represents NMFS' opinion on the effects of these actions on endangered and threatened species and critical habitat that has been designated for those species. A complete record of this consultation is on file at NMFS Office of Protected Resources in Silver Spring, Maryland.

1.1 Background

The NSF is proposing to fund a seismic survey for October 2015 in the eastern Mediterranean Sea around Greek islands near Santorini and in the Hellenic Trench (near the island of Crete). In conjunction with this action, the NMFS Permits and Conservation Division would issue an IHA under the MMPA for marine mammal takes that could occur during the NSF seismic survey. This document represents NMFS's ESA Interagency Cooperation Division's Opinion on the effects of the two proposed federal actions on threatened and endangered species, and has been prepared in accordance with section 7 of the ESA.

1.2 Consultation History

On April 16, 2015, the NMFS' ESA Interagency Cooperation Division received a request for formal consultation pursuant to section 7 of the ESA from the NSF to incidentally harass marine mammal and sea turtle species during the seismic survey. On the same date, the NMFS' Permits and Conservation Division received an application from the L-DEO to incidentally harass marine mammal species pursuant to the MMPA during the proposed seismic survey.

On June 25, 2015, NSF sent GIS files representing the survey tracklines around Santorini, which had changed from what was presented in the April 16th request.

On July 23, 2015, the NMFS' ESA Interagency Cooperation Division met with the NMFS' Permits and Conservation Division and NSF to discuss the timing of the seismic survey. NSF stated that the seismic survey schedule would be changing, and was now proposed to take place in mid-November through December. A revised draft Environmental Analysis was received on August 24, 2015. Information was sufficient to initiate consultation with the NSF on this date.

On August 3, 2015, the NMFS' ESA Interagency Cooperation Division received a request for formal consultation under section 7 of the ESA from the NMFS' Permits and Conservation Division.

On September 4, 2015, the NMFS' Permits and Conservation Division sent the application for the proposed seismic survey out to reviewers and published a notice in the *Federal Register* soliciting public comment on their intent to issue an IHA.

This opinion is based on information provided in the:

- MMPA IHA application
- draft public notice of proposed IHA
- a draft environmental assessment prepared pursuant to the National Environmental Policy Act
- 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.

2 DESCRIPTION OF THE PROPOSED ACTION

"Action" means all activities or programs of any kind authorized, funded, or carried out, in whole or in part, by federal agencies. Two federal actions were evaluated in this Opinion. The first is the NSF's proposal to allow the use of its research vessel, *Marcus G. Langseth* (*Langseth*), operated by the L-DEO, to conduct a seismic survey in the eastern Mediterranean Sea around Greece in November and December of 2015, in support of an NSF-funded collaborative research project led by the University of Oregon. The second is the NMFS' Permits and Conservation Division proposal to issue an IHA authorizing non-lethal "takes" by Level B harassment pursuant to section 101 (a)(5)(D) of the MMPA. The information presented here is based primarily upon the Environmental Analysis provided by NSF as part of the initiation package.

2.1 National Science Foundation Proposed Action

The NSF proposes to allow the use of its research vessel, the *Langseth*, to conduct a seismic survey on and around the Greek island of Santorini and in the Hellenic Trench in the eastern Mediterranean during an approximate 29 day period from mid-November through December 2015. An array of thirty-six airguns will be deployed as an energy source.

There are two portions of the proposed activity focusing on two separate areas: a grid of transect lines to be surveyed surrounding the island of Santorini, and a single transect line to be surveyed in the Hellenic trench subduction zone to the northwest/southwest of Crete (Figure 4).

For the Santorini portion of the seismic survey, the receiving system would consist of 93 ocean bottom seismometers (OBSs) and 30 land seismometers on the islands of Santorini, Anafi and Therasia. For the Hellenic Trench portion of the seismic survey, a single 8 km hydrophone streamer would be used. In addition, a multibeam echosounder (MBES) and a sub-bottom profiler (SBP) will continuously operate from the *Langseth,* except during transit to the survey site.

The purpose of the proposed activities is to collect data on and around the island of Santorini to examine the crustal magma plumbing of the volcanic system. The high-resolution 3D seismic refraction survey would provide valuable geologic information on the structure of the volcano system. By studying the Hellenic Trench subduction zone, scientists could image the megathrust fault between Peloponnesus and Crete to better understand seismic activity in this zone.

2.1.1 Schedule

The NSF proposes to allow the use of the *Langseth* by L-DEO during roughly 16 days of seismic operations and an additional 13 days of non-airgun operations (e.g., OBS deployment and retrieval, hydrophone streamer deployment and retrieval). Some minor deviation from the proposed dates is possible, depending on logistics, weather conditions, and the need to repeat some survey lines if data quality is substandard. During an approximate 29-day period starting in mid-November through December 2015, corresponding to an effective IHA, the *Langseth* would survey the action area (Figure 4). The *Langseth* would depart from Piraieus, Greece, and return to Iraklio, Crete.

2.1.2 Source Vessel Specifications

The *Langseth* will tow a source 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 about 8.3 km/h (4.5 knots). When not towing seismic survey gear, the *Langseth* typically cruises at 18.5 km/h (10 knots). No chase vessel will be used during operations.

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

2.1.3 Airgun Description

The airgun array will consist of the full airgun array of four strings with 36 airguns, plus four spares (**Table 1**). The total discharge volume will be 6,600 in^3. The airgun configuration includes four of linear arrays or "strings". Each string will have ten airguns. Up to nine airguns in one string would fire at any one time. The four-airgun strings will be towed approximately 150 m behind the vessel. The tow depth of the array will be 9 or 12 m. The airgun array will fire roughly every 35 to 170 seconds (or approximately every 80-390 m) for OBS lines and about every 22 seconds (~50 m) for the multi-channel seismic lines with the streamer in the Hellenic Trench portion of the study. During firing, a brief (approximately 0.1 s) pulse of sound will be emitted, but will be silent during the intervening periods. This signal attenuates as it moves away from the source, decreasing in amplitude, but also increasing in signal duration. Airguns will operate continually during the survey period except for unscheduled shutdowns.

Table 1 Specifications of the source array to be used by the R/V *Langseth* during the proposed seismic activities.

Source array specifications	
Energy source	20 Bolt 1500LL airguns @ 180-360 in$_3$ 16 Bolt 1900LLX airguns @ 40-120 in$_3$ 36-1,950 psi bolt airguns of 120-220 in^3 each, in four strings of nine operating airguns per string

Source our tput (downward)-4 airgun array	0-pk = 259 dB re 1 μPa-m pk-pk = 265 dB re 1 μPa-m or 0-pk = 258 dB re 1 μPa-m pk-pk = 264 dB re 1 μPa-m 0-pk is 240.4 dB re 1 μPa·m; pk-pk is 246.3-246.7 dB re 1 μPa·m
Air discharge volume	~ 6,600 in^3
Dominant frequency components	0–188 Hz

Because the actual source originates from thirty-six 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.

2.1.4 OBS and Land-based Seismometers

To conduct the Santorini portion of the study, the *Langseth* would deploy a total of 93 OBSs before beginning the tracklines around Santorini; the OBSs would be recovered after the survey is completed. OBSs from two sources would be used during the proposed activities—30 from Woods Hole Oceanographic Institution and 63 from Scripps Institution of Oceanography. The OBSs differ slightly in their configurations (Table 2). Ocean bottom seismometers would not be used in the Hellenic Trench portion of the seismic activities.

Table 2 Specificiations for the OBSs to be used during the proposed seismic activities around the island of Santorini.

Dimensions	Woods Hole Oceanographic Institution OBS	Scripps Instituion of Oceanography OBS
Height	1 m	0.9 m
Diameter	50 cm	97 cm
Anchor Description	Hot-rolled steel; 23kg	Iron grates; 36kg
Anchor Dimensions	2.5 x 30.5 x 38.1 cm	7 x 91 x 91.5 cm

To retrieve an OBS, an acoustic release transponder activates the instrument at a frequency of 8-11 kHz, and the receiver detects the response at a frequency of 11.5-13 kHz, at which point the burn-wire releases the instrument from the anchor and the devices floats to the surface.

There are currently 21 permanent land-based seismometers on the study area—20 on Santorini, and one on Anáfi. To conduct the study, an additional nine land-based seismometers would be installed on Santorini by digging a small trench and placing equipment there. These land-based seismometers would be removed once the study was complete.

2.1.5 Multibeam Echosounder and Sub-bottom Profiler

Along with airgun operations, two additional acoustical data acquisition systems will operate during the survey from the *Langseth*. The multibeam echosounder and sub-bottom profiler systems will map the ocean floor during the survey. 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 μPa·m_{rms}. For deepwater operation, each "ping" consists of eight successive fan-shaped transmissions, each 2 to 15 ms in duration and each ensonifying a sector that extends 1° fore–aft. The eight successive transmissions span an overall cross-track angular extent of about 150°, with 2 ms gaps between the pulses for successive sectors (Maritime 2005).

The sub-bottom profiler provides information about the sedimentary features and the bottom topography that is being mapped simultaneously by the multibeam echosounder (**Table 3**). The output varies with water depth from 50 watts in shallow water to 1,000 (204 dB) watts in deep water. The pulse interval is 1 s, but a common mode of operation is to broadcast five pulses at 1-s intervals followed by a 5-s pause.

Table 3 Sub-bottom profiler specifications of the R/V *Langseth.*

***Langseth* sub-bottom profiler specifications**	
Maximum/normal source output (downward)	204 dB re 1 μPa·m; 800 watts
Dominant frequency component	3.5 kHz
Bandwidth	1.0 kHz with pulse duration 4 ms
	0.5 kHz with pulse duration 2 ms
	0.25 kHz with pulse duration 1 ms

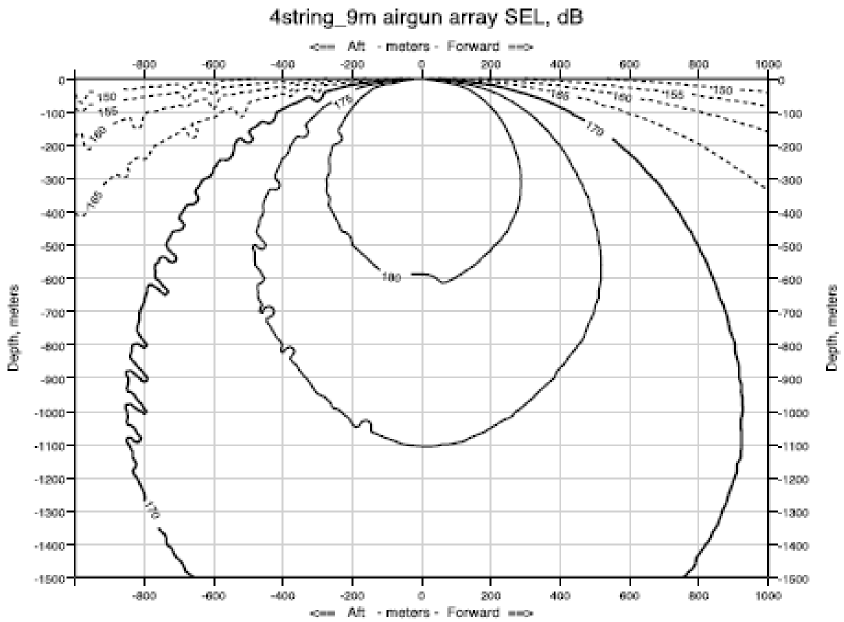
Nominal beam width	30º
Pulse duration	1, 2, or 4 ms

2.1.6 Proposed Exclusion Zones

The NSF identifies in its EA that the L-DEO will implement exclusion zones (EZs) 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 EZs are based upon modeled sound levels at various distances from the *Langseth*, described below.

2.1.6.1 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 a 36-airgun array (Figure 1 and Figure 2) as well as a 40 in^3 single 1900LLX airgun used during power-downs (Figure 3). In shallow water, empirical data concerning 180 and 160 dB re 1 μPa_{rms} distances were acquired during the acoustic calibration study of the *Langseth*'s 36-airgun 6,600 in^3 array in 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 or 12 m tow depth). To adapt the shallow-water measurements obtained during the calibration survey to the proposed tow depth(s), scaling factors were applied to the distances reported by Diebold et al. (2009) for shallow waters, and this scaling is done according to the sound exposure level (SEL) contours obtained from the free-field modeling. The deep-water EZ radii (>1000m) were obtained from L-DEO model results (to a water depth of 2000m). The EZ radii for intermediate water depths (100-1000m) were derived from the deep-water ones by applying a 1.5 correction factor. Figures 1-3 show predicted distances of the various configurations of the airguns.



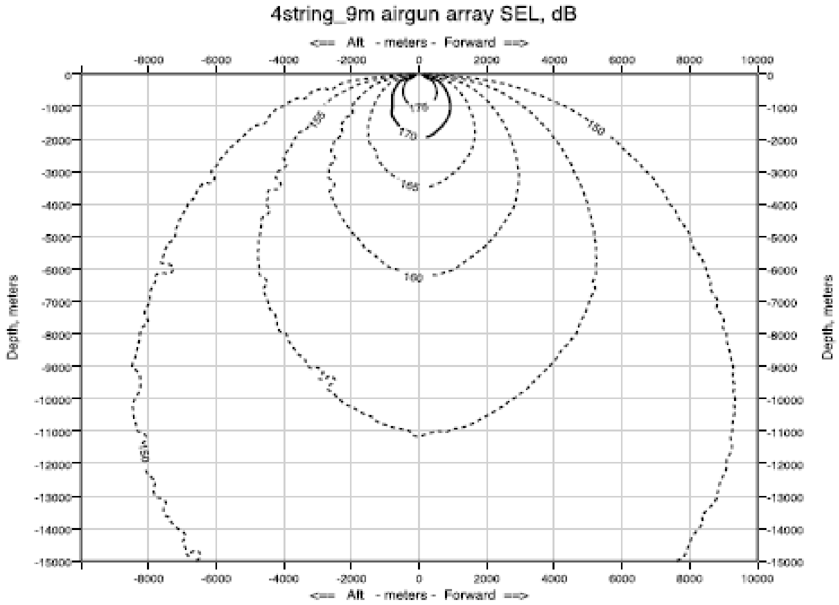
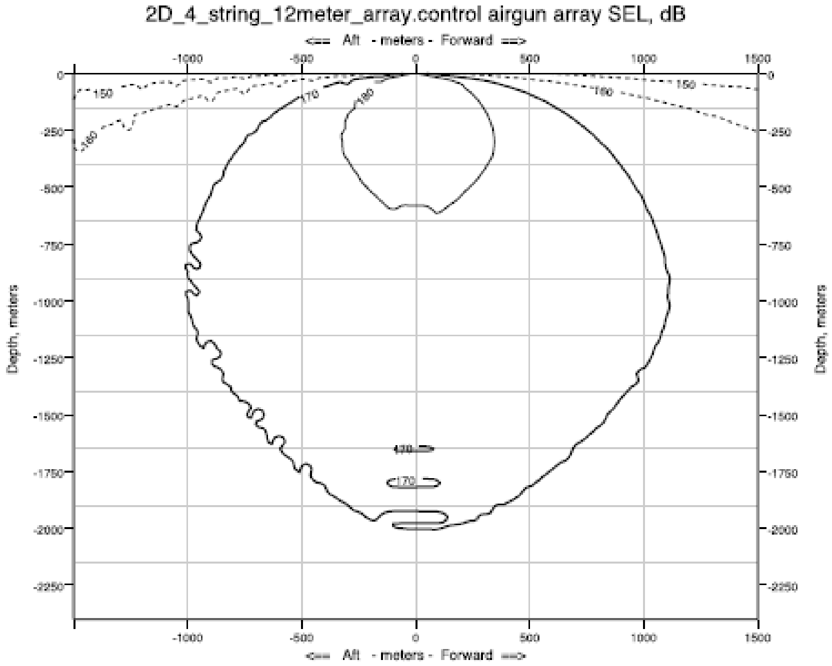


Figure 1 Modelled distances for the 36-airgun array at nine meter tow depth in deep water; top plot provides the radius to the 170-dB SEL isopleth, bottom plot provides the radius to the 150-dB SEL isopleth. From LGL 2015.


2D_4_string_12meter_array.control airgun array SEL, dB
<== Aft - meters - Forward ==>
-1000
-500
0
500
1000
1500
Depth, meters
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-250
-500
-750
-1000
-1250
-1500
-1750
-2000
-2250
150
160
170
180

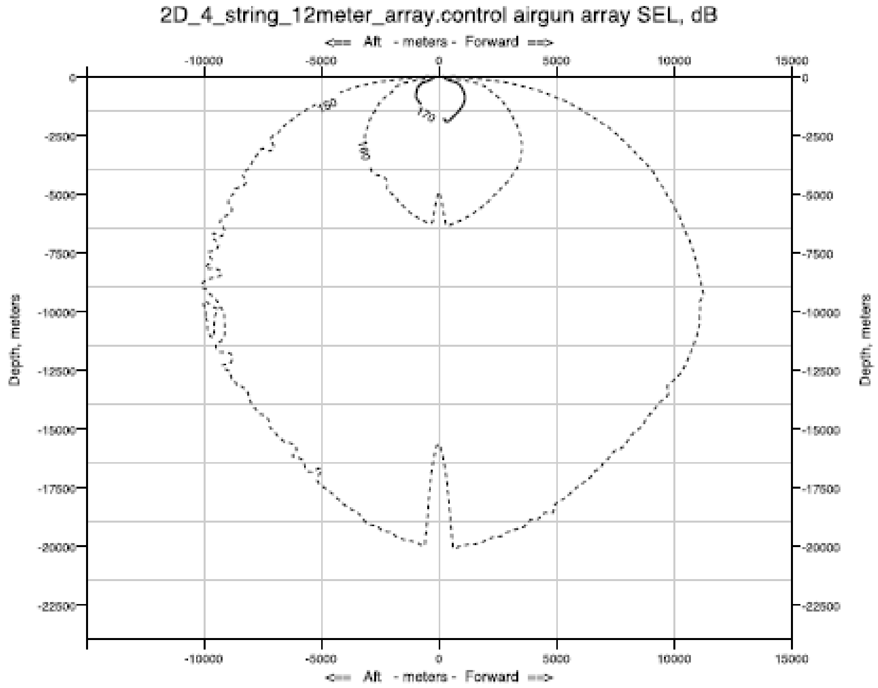


Figure 2 Modelled distances for the 36-airgun array at twelve meter tow depth in deep water; top plot provides the radius to the 170-dB SEL isopleth, bottom plot provides the radius to the 150-dB SEL isopleth. From LGL 2015.

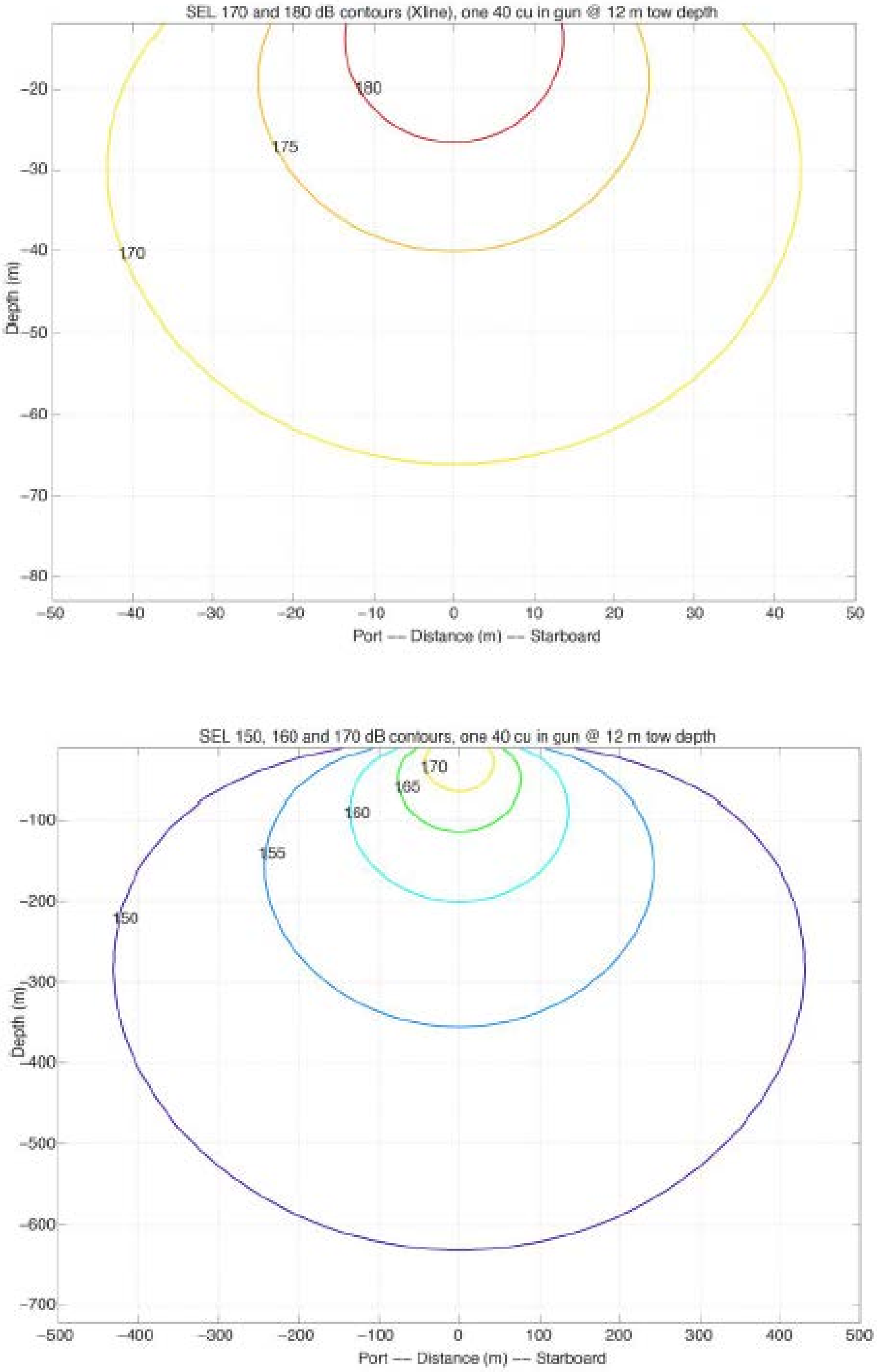


Figure 3 Modelled distances for the single 40-in^3 airgun array at twelve meter tow depth in deep water; top plot provides the radius to the 170-dB SEL isopleth, bottom plot provides the radius to the 150-dB SEL isopleth. From LGL 2015.

Table 4 shows the distances at which four rms (root mean squared) sound levels are expected to be received from the 36-airgun arrays and a single airgun at different depth strata. Because the proposed action would take place in an area with variable depths, NSF provided predicted distances to which sound levels could be received at different depths. The 180 dB re 1 μPa_{rms} distance is the safety criteria as specified by NMFS (1995) as applicable to cetaceans under the MMPA. The 180 dB will be used as the exclusion zone (EZ) for marine mammals, as required by NMFS during most other recent L-DEO seismic projects (Holst and Beland 2008; Holst and Smultea 2008b; Holst et al. 2005a; Holt 2008; Smultea et al. 2004). The 180 dB isopleth would also be the EZ 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 4 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 under 100 m, 100-1000 m, and >1000m. Adapted from LGL 2015.

Source, volume, and tow depth	**Water Depth (m)**	**Predicted RMS radii (m)**		
		180 dB	**166 dB**	**160 dB**
36-airgun array 6600 in^3 @ 9 m	>1000 m	927	3,740	5,780
	100-1000 m	1,391	5,610	8,670
	<100 m	2,060	10,862	22,580
36-airgun array 6600 in^3 @ 12 m	>1000 m	1,116	4,411	6908
	100-1000 m	1,647	6,617	10,362
	<100 m	2,480	12,630	27,130
single Bolt airgun, 40 in^3 @ 9 or 12 m	>1000 m	96	216	431
	100-1000 m	100	324	647
	<100 m	100	501	1,041

2.2 NMFS Permits and Conservation Division's Incidental Harassment Authorization

The NMFS' Permits and Conservation Division is proposing to issue an IHA authorizing non-lethal "takes" by Level B harassment of marine mammals incidental to the planned seismic survey. The IHA will be valid from October 13, 2015 through December 31, 2015. The Permits and Conservation Division proposes to issue the IHA by October 13, 2015 so that NSF will have the IHA in hand prior to October 19, 2015, the proposed sail date for the *Langseth's* trans-Atlantic voyage. The IHA will authorize the incidental harassment of the following endangered species: Mediterranean monk seals (*Monachus monachus*), fin whales (*Balaenoptera physalus*), humpback whales (*Megaptera novaeangliae*), sei whales (*Balaenoptera borealis*), sperm whales (*Physeter macrocephalus*), and other marine mammals listed under the Marine Mammal Protection Act. The proposed IHA identifies the following requirements that L-DEO must comply with as part of its authorization.

- Visual Observers
    - Utilize two, National Marine Fisheries Service-qualified, vessel-based Protected Species Visual Observers (visual observers) to watch for and monitor marine mammals near the seismic source vessel during daytime airgun operations (from civil twilight-dawn to civil twilight-dusk) and before and during start-ups of airguns day or night.
        - At least one visual observer will be on watch during meal times and restroom breaks.
        - Observer shifts will last no longer than four hours at a time.
        - Visual observers will also conduct monitoring while the *Langseth* crew deploy and recover the airgun array and streamers from the water.
        - When feasible, visual observers will conduct observations during daytime periods when the seismic system is not operating for comparison of sighting rates and behavioral reactions during, between, and after airgun operations.
    - The *Langseth*'s vessel crew will also assist in detecting marine mammals, when practicable. Visual observers will have access to reticle binoculars (7x50 Fujinon) and big-eye binoculars (25x150).
- Exclusion Zones
    - Establish a 180-decibel (dB) or 190-dB exclusion zone for cetaceans and pinnipeds, respectively, before starting the airgun array; and a 180-dB or 190 dB exclusion zone for cetaceans and pinnipeds, respectively for the single airgun. Observers will use the predicted radius distance for the 180-dB or 190-dB exclusion zones for cetaceans and pinnipeds.
- Visual Monitoring at the Start of Airgun Operations

  - Monitor the entire extent of the exclusion zones for at least 30 minutes (day or night) prior to the ramp-up of airgun operations after a shutdown.
  - Delay airgun operations if the visual observer sees a cetacean within the 180-dB exclusion zones for cetaceans or 190-dB exclusion zone for pinnipeds until the marine mammal(s) has left the area.
  - If the visual 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, he/she should assume that the animal has moved beyond the 180-dB exclusion zone for cetaceans or 190-dB exclusion zone for pinnipeds.
  - If for any reason the visual observer cannot see the full 180-dB exclusion zone for cetaceans or the 190-dB exclusion zone for pinnipeds for the entire 30 (i.e., rough seas, fog, darkness), or if marine mammals are near, approaching, or within zone, the *Langseth* may not resume airgun operations.

- Passive Acoustic Monitoring
  - Utilize the passive acoustic monitoring (PAM) system, to the maximum extent practicable, to detect and allow some localization of marine mammals around the *Langseth* during all airgun operations and during most periods when airguns are not operating. One visual observer and/or bioacoustician will monitor the PAM at all times in shifts no longer than 6 hours. A bioacoustician will monitor the PAM at all times in shifts no longer than 6 hours. 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.
  - Do and record the following when an observer detects an animal by the PAM:
    - Notify the visual observer immediately of a vocalizing marine mammal so a power-down or shut-down can be initiated, if required;
    - Enter the information regarding the vocalization into a database. The data to be entered include an acoustic encounter identification number, whether it was linked with a visual sighting, date, time when first and last heard and whenever any additional information was recorded, position, and water depth when first detected, bearing if determinable, species or species group (e.g., unidentified dolphin, sperm whale), types and nature of sounds heard (e.g., clicks, continuous, sporadic, whistles, creaks, burst pulses, strength of signal, etc.), and any other notable information.

- Ramp-up Procedures
    - Implement a “ramp-up” procedure when starting the airguns at the beginning of seismic operations or any time after the entire array has been shutdown, 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-minute period.
    - During ramp-up, the observers will monitor the exclusion zone, and if marine mammals are sighted, a course/speed alteration, power-down, or shutdown will be implemented as though the full array were operational.
- Recording Visual Detections
    - Visual observers must record the following information when they have sighted a marine mammal:
        - Species, group size, age/size/sex categories (if determinable), behavior when first sighted and after initial sighting, heading (if consistent), bearing and distance from seismic vessel, sighting cue, apparent reaction to the airguns or vessel (e.g., none, avoidance, approach, paralleling, etc. and including responses to ramp-up), and behavioral pace; and
        - Time, location, heading, speed, activity of the vessel (including number of airguns operating and whether in a state of ramp-up or shutdown), Beaufort sea state and wind force, visibility, and sun glare; and
        - The data listed 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.
- Speed or Course Alteration
    - Alter speed or course during seismic operations if a marine mammal, based on its position and relative motion, appears likely to enter the relevant exclusion zone. If speed or course alteration is not safe or practicable, or if after alteration the marine mammal still appears likely to enter the exclusion zone, the Holder of this Authorization will implement further mitigation measures, such as a shutdown.
- Power-Down Procedures
    - Power down the airguns if a visual observer detects a marine mammal within, approaching, or entering the relevant exclusion zones. A power-down means

reducing the number of operating airguns to a single operating 40 in^3 airgun. This would reduce the exclusion zone to the degree that the animals(s) is outside of it.

- Resuming Airgun Operations after a Power-Down
  - Following a power-down, if the marine mammal approaches the smaller designated exclusion zone, the airguns must then be completely shut-down. Airgun activity will not resume until the observer has visually observed that the marine mammal(s) exiting the exclusion zone and is not likely to return, or has not been seen within the exclusion zone for 15 minutes for species with shorter dive durations (small odontocetes) or 30 minutes for species with longer dive durations (mysticetes and large odontocetes, including sperm, pygmy sperm, dwarf sperm, killer, and beaked whales).
  - Following a power-down and subsequent animal departure, the *Langseth* may resume airgun operations at full power. Initiation requires that the observers can effectively monitor the full exclusion zones described in Condition 6(b). If the observer sees a marine mammal within or about to enter the relevant zones then the *Langseth* will implement a course/speed alteration, power-down, or shutdown.
- Shutdown Procedures
  - Shutdown the airgun(s) if a visual observer detects a marine mammal within approaching, or entering the relevant exclusion zone. A shutdown means that the *Langseth* turns off all operating airguns.
  - If a Mediterranean monk seal 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 minutes after the last documented visual sighting.
- Resuming Airgun Operations after a Shutdown
  - Following a shutdown, if the observer has visually confirmed that the animal has departed the 180-dB zone for cetaceans or the 190-dB zone pinnipeds within a period of less than or equal to 8 minutes after the shutdown, then the *Langseth* may resume airgun operations at full power.
  - If the observer has not seen the animal depart the 180-dB zone for cetaceans or the 190-dB zone for pinnipeds, the *Langseth* shall not resume airgun activity until 15 minutes has passed for species with shorter dive durations (small odontocetes) or 30 minutes for species with longer dive durations (mysticetes and large

odontocetes, including sperm, pygmy sperm, dwarf sperm, killer, and beaked whales). The *Langseth* will follow the ramp-up procedures described earlier.

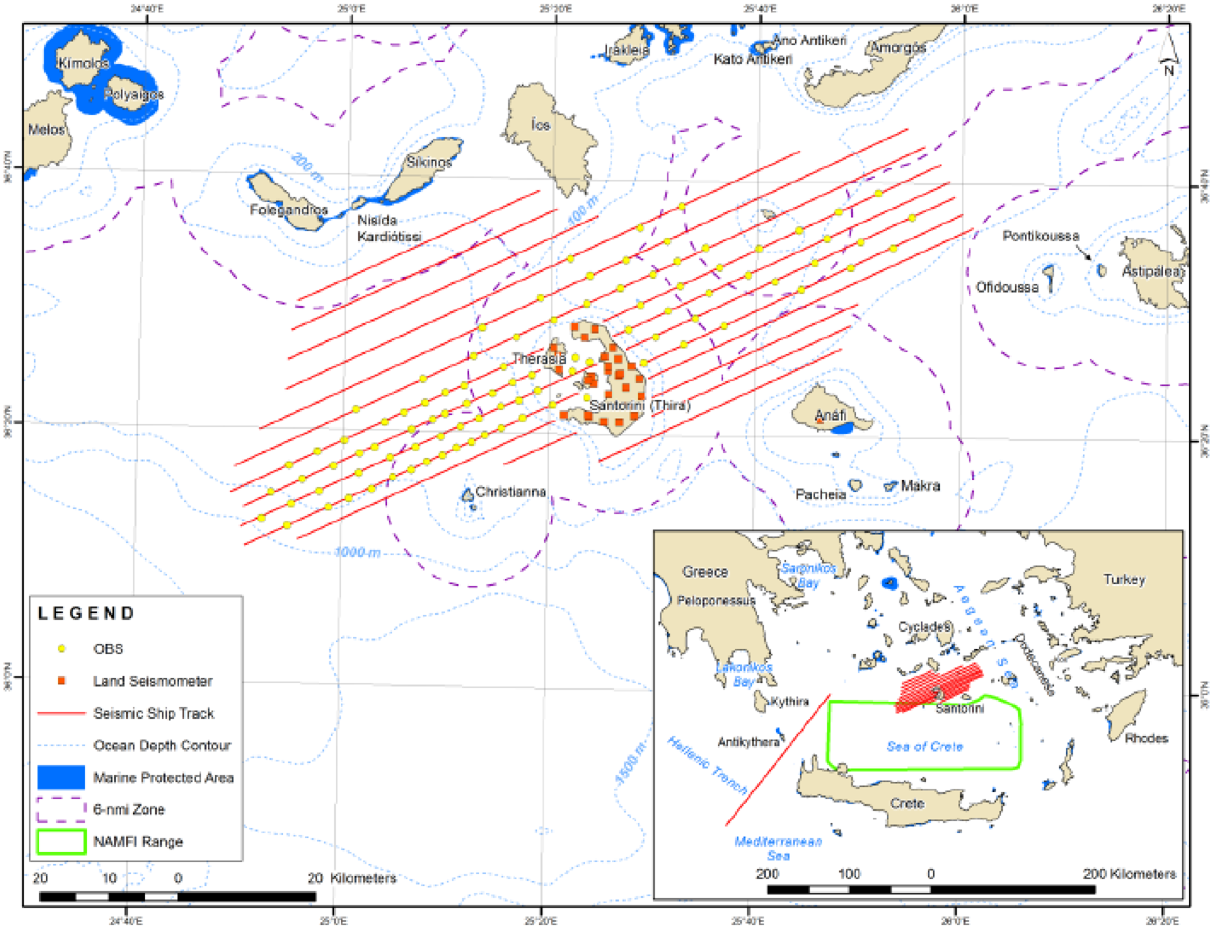
- Survey Operations at Night
  - The *Langseth* may continue marine geophysical surveys into the night and low-light hours if the Holder of the Authorization initiates these segment(s) of the survey when the observers can view and effectively monitor the full relevant exclusion zones.
  - This Authorization does not permit the Holder of this Authorization to initiate airgun array operations from a shut-down position at night or during low-light hours (such as in dense fog or heavy rain) when the visual observers cannot view and effectively monitor the full relevant exclusion zones.
  - To the maximum extent practicable, the Holder of this Authorization should schedule seismic operations (i.e., shooting the airguns) during daylight hours.
- Mitigation Airgun
  - The *Langseth* may operate a small-volume airgun (i.e., mitigation airgun) during turns and maintenance at approximately one shot per minute. The *Langseth* would not operate the small-volume airgun for longer than three hours in duration during turns. During turns or brief transits between seismic tracklines, one airgun would continue to operate.
- Special Procedures for Large Whale Concentrations
  - The *Langseth* will power-down the array and avoid concentrations of fin (*Balaenoptera physalus*) and/or sperm whales (*Physeter macrocephalus*) if possible (i.e., avoid exposing concentrations of these animals to sounds greater than 160-dB re: 1 μPa).
  - For purposes of the survey, a concentration or group of whales will consist of six or more individuals visually sighted that do not appear to be traveling (e.g., feeding, socializing, etc.). The *Langseth* will follow the procedures described earlier for resuming operations after a power down.
  - Restrict the operation of the multi-beam echosounder and sub-bottom profiler during transit to and from the survey area.

2.3 Action Area

Action area means all areas affected directly, or indirectly, by the Federal action, and not just the immediate area involved in the action (50 CFR 402.02).

The proposed action will occur in two locations near Greece in the eastern Mediterranean Sea: 1) around Santorini Island in the Aegean Sea, and 2) across the Hellenic Trench subduction zone near Crete (Figure 4).

Figure 4 Location of the proposed seismic survey in the eastern Mediterranean Sea. From LGL 2015.



2.3.1 Schedule

To correspond with NSF's proposed seismic survey (section 2.1.1), the NMFS' Permits and Conservation Division proposes to issue an authorization that is effective from November 19, 2015 to December 31, 2015.

2.3.2 Greek Territorial Seas and the Action Area

Greece considers its territorial seas to extend out 6 nautical miles (n mi). According to the draft Environmental Analysis that NSF prepared for this action, approximately half of the 1,258 km of survey lines around Santorini will take place inside the 6 n mi line. For the single, long transect line to be shot in the Crete/Hellenic Trench subduction zone, about 92% of the line is located outside the 6 n mi. territorial seas line.

In its draft Environmental Analysis, NSF presented the action area in terms of the Greek territorial seas—that is, 6 n mi. A nation's territorial seas is the sovereign territory of that country. A country may claim up to 12 n mi as territorial seas, but Greece has declared its territorial seas boundary at 6 n mi.

NMFS' jurisdiction under the ESA and MMPA only applies to the portions of the seismic survey which occur outside the 6 n mi boundary.

The fact that portions of the proposed action fall both inside and outside of the 6 n mi boundary (the high seas under the ESA) presents us with a complicated situation. For ESA section 7 consultations, we are required to examine the effects of the action throughout the entire action area in making our jeopardy determination. However, we do not have authority under the ESA to authorize incidental take or not within the sovereign territory of Greece (i.e., within 6 n mi).

The ESA defines action area as "all areas to be affected directly or indirectly by the Federal action and not merely the immediate area involved in the action." Although portions of the tracklines do not occur in the high seas (where the ESA has explicit jurisdiction), we are obligated to consider the effects of the action throughout the entire action area. Therefore, we must consider the 6 n mi boundary in relation to:

- The location of the tracklines, and
- The extent of the ensonified area.

The ensonified area refers to the predicted distances to which sound levels are expected to be received. In the draft EA, NSF presents the predicted distances to which sound levels are expected to occur for different decibel levels and at different water depths. Generally, sound travels farther in shallow water than it does in deep water.

It is possible that tracklines that are technically located within the 6 n mi boundary could, during the seismic survey, emit an ensonified area that goes beyond 6 n mi. Water depth will play a role as well, because the predicted distances to which sound can travel changes with depth. The water depths around the Santorini area range from 20-500m (inside the 6 n mi boundary), and 500-1,000m in the high seas. The Crete survey line takes place in deeper waters; the water depths in the northern part of the trackline range from 500-1,000m deep, and on the southern part of the line, from 1,000-2,000 m deep (Figure 4).

This opinion is going to consider two exposure scenarios to fulfill our requirements under the ESA:

1. Estimate exposure to determine the effects of the proposed action throughout the entire action area (inside and outside the 6 n mi boundary) to reach the jeopardy determination, and
2. Estimate exposure in the portions of the action area where NMFS has jurisdiction under the ESA (to estimate numbers of allowed take for an incidental take statement).

2.4 Interrelated and Interdependent Actions

Interrelated actions are those that are part of a larger action and depend on that action for their justification. *Interdependent* actions are those that do not have independent use, apart from the action under consideration.

The two proposed actions considered in this opinion are interdependent. The Permits and Conservation Division's proposal to issue an MMPA authorization is interdependent on NSF's proposed seismic activities, as it would not have an independent use if not for the actual activity NSF proposed. Likewise, NSF's proposed action would not carry forward without the authorization to exempt marine mammal take from the Permits and Conservation Division.

3 OVERVIEW OF NMFS' ASSESSMENT FRAMEWORK

Section 7 (a)(2) of the ESA requires Federal agencies, in consultation with NMFS, to insure that their actions either are not likely to jeopardize the continued existence of endangered or threatened species; or adversely modify or destroy their designated critical habitat.

"To jeopardize the continued existence of an ESA-listed species" means to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of an ESA-listed species in the wild by reducing the reproduction, numbers, or distribution of that species (50 CFR §402.02). The jeopardy analysis considers both survival and recovery of the species.

This biological opinion includes both a jeopardy analysis and an adverse modification analysis.

The jeopardy analysis relies upon the regulatory definition of "to jeopardize the continued existence of a listed species," which is "to engage in an action that would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species" (50 CFR 402.02). Therefore, the jeopardy analysis considers both survival and recovery of the species.

The adverse modification analysis considers the impacts on the conservation value of designated critical habitat. This biological opinion does not rely on the regulatory definition of "destruction or adverse modification" of critical habitat at 50 C.F.R. 402.02, which was invalidated by Gifford Pinchot Task Force v. USFWS, 378 F.3d 1059 (9th Cir. 2004), amended by 387 F.3d 968 (9th Cir. 2004). Instead, we have relied upon the statutory provisions of the ESA to complete our analysis with respect to critical habitat.

Section 7 assessment involves the following steps:

1) We identify the proposed action and those aspects (or stressors) of the proposed action that are likely to have direct or indirect effects on the physical, chemical, and biotic environment within the action area, including the spatial and temporal extent of those stressors.

2) We identify the ESA-listed species and designated critical habitat that are likely to co-occur with those stressors in space and time.

3) We describe the environmental baseline in the action area including: past and present impacts of Federal, state, or private actions and other human activities in the action area; anticipated impacts of proposed Federal projects that have already undergone formal or early section 7 consultation, impacts of state or private actions that are contemporaneous with the consultation in process.

4) We identify the number, age (or life stage), and gender of ESA-listed individuals that are likely to be exposed to the stressors and the populations or subpopulations to which those individuals belong. This is our exposure analysis.

5) We evaluate the available evidence to determine how those ESA-listed species are likely to respond given their probable exposure. This is our response analyses.

6) We assess the consequences of these responses to the individuals that have been exposed, the populations those individuals represent, and the species those populations comprise. This is our risk analysis.

7) The adverse modification analysis considers the impacts of the proposed action on the critical habitat features and conservation value of designated critical habitat. This opinion does not rely on the regulatory definition of “destruction or adverse modification” of critical habitat at 50 C.F.R. 402.02. Instead, we have relied upon the statutory provisions of the ESA to complete the following analysis with respect to critical habitat.[1]

8) We describe any cumulative effects of the proposed action in the action area.

   Cumulative effects, as defined in our implementing regulations (50 CFR §402.02), are the effects of future state or private activities, not involving Federal activities, that are reasonably certain to occur within the action area. Future Federal actions that are unrelated to the proposed action are not considered because they require separate section 7 consultation.

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[1] Memorandum from William T. Hogarth to Regional Administrators, Office of Protected Resources, NMFS (Application of the “Destruction or Adverse Modification” Standard Under Section 7(a)(2) of the Endangered Species Act) (November 7, 2005).

9) We integrate and synthesize the above factors by considering the effects of the action to the environmental baseline and the cumulative effects to determine whether the action could reasonably be expected to:

    a) Reduce appreciably the likelihood of both survival and recovery of the ESA-listed species in the wild by reducing its numbers, reproduction, or distribution; or

    b) Reduce the conservation value of designated or proposed critical habitat. These assessments are made in full consideration of the status of the species and critical habitat.

10) We state our conclusions regarding jeopardy and the destruction or adverse modification of critical habitat.

If, in completing the last step in the analysis, we determine that the action under consultation is likely to jeopardize the continued existence of ESA-listed species or destroy or adversely modify designated critical habitat, we must identify a reasonable and prudent alternative (RPA) to the action. The RPA must not be likely to jeopardize the continued existence of ESA-listed species nor adversely modify their designated critical habitat and it must meet other regulatory requirements.

To comply with our obligation to use the best scientific and commercial data available, 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' Permits and Conservation Division when it initiates formal consultation, the general scientific literature, and our expert opinion.

To identify information relevant to the potential stressors and responses of marine mammals, sea turtles, and fish species that may be affected by the proposed action to draw conclusions about the likely risks to the continued existence of these species and the conservation value of their critical habitat during the consultation, we conducted electronic searches of the general scientific literature using search engines, including 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' 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 listed species when, in fact, such adverse effects are likely (i.e., Type II error).

3.1 Assessment Approach applied in this consultation

In this particular assessment, we identified the potential stressors associated with the action and determined which were probable 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, where those individuals may respond with behaviors that may result in fitness consequences (Francis and Barber 2013; Nowacek and Tyack 2013) However, it should not be assumed that anthropogenic stressors lead to fitness consequences at the individual or population levels (New et al. 2013).

In order to reach conclusions regarding whether proposed actions are likely to jeopardize ESA-listed species, we had to make several assumptions. These included:

- Baleen whales can generally hear low-frequency sound (Southall et al. 2007b) better than high frequencies (Southall et al. 2007b), as the former is primarily the range in which they vocalize. Humpback whales frequently vocalize with mid-frequency sound (Southall et al. 2007b) 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., fin, North Atlantic right, and sei whales) and those that hear at low- to mid-frequencies (blue and 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 at different received sound levels will respond similarly to their close taxonomic or ecological relatives (i.e., baleen whales respond similarly to each other; same for sea turtles).

4 STATUS OF ESA-LISTED SPECIES

This section identifies the ESA-listed species that potentially occur within the action area (Figure 4) that may be affected by the proposed seismic activities. It then summarizes the biology and ecology of those species and what is known about their life histories in the action area. The species potentially occurring within the action area are ESA-listed in Table 5, along with their regulatory status.

Table 5. Threatened and endangered species that may be affected by NSF's proposed action of seismic activities in the eastern Mediterranean Sea.

Species	ESA Status	Critical Habitat	Recovery Plan
Marine Mammals – Cetaceans			
Fin Whale (*Balaenoptera physalus)*	E – 35 FR 18319	-- --	75 FR 47538
Humpback Whale (Megaptera novaeangliae)	E – 35 FR 18319	-- --	55 FR 29646
Sei Whale (Balaenoptera borealis)	E – 35 FR 18319	-- --	76 FR 43985
Sperm Whale (*Physeter macrocephalus*)	E – 35 FR 18319	-- --	75 FR 81584
Marine Mammals – Pinnipeds			
Mediterranean Monk Seal (*Monachus monachus*)	E – 35 FR 8491	-- --	-- --
Sea Turtles			
Green Turtle (*Chelonia mydas*)	E – 43 FR 32800	63 FR 46693	63 FR 28359
Leatherback Turtle (*Dermochelys coriacea*)	E – 61 FR 17	44 FR 17710	63 FR 28359
Loggerhead Turtle (*Caretta caretta*) – Mediterranean Sea DPS	E – 76 FR 58868	-- --	63 FR 28359
Fishes			
Scalloped Hammerhead Shark (Sphyrna lewini) – Eastern Atlantic DPS	E – 79 FR 38213	-- --	-- --
Adriactic sturgeon (*Acipenser naccarii*)	E – 79 FR 31222	-- --	-- --
European Sturgeon (*Acipenser sturio*)	E – 79 FR 31222	-- --	-- --

4.1 ESA-listed Species and Critical Habitat Not Likely to be Adversely Affected

NMFS uses two criteria to identify the ESA-listed or critical habitat that are not likely to be adversely affected by the proposed action, as well as the effects of activities that are interrelated to or interdependent with the Federal agency's proposed action. The first criterion is exposure, or some reasonable expectation of a co-occurrence, between one or more potential stressors associated with the proposed activities and ESA-listed species or designated critical habitat. If we conclude that an ESA-listed species or designated critical habitat is not likely to be exposed to the proposed activities, we must also conclude that the species or critical habitat is not likely to be adversely affected by those activities.

The second criterion is the probability of a response given exposure. ESA-listed species or designated critical habitat that is exposed to a potential stressor but is likely to be unaffected by the exposure is also not likely to be adversely affected by the proposed action. We applied these criteria to the species ESA-listed in Table 5 and we summarize our results below.

An action warrants a "may affect, not likely to be adversely affected" finding when its effects are wholly *beneficial, insignificant* or *discountable*. *Beneficial* effects have an immediate positive effect without any adverse effects to the species or habitat. Beneficial effects are usually discussed when the project has a clear link to the ESA-listed species or its specific habitat needs and consultation is required because the species may be affected.

Insignificant effects relate to the size or severity of the impact and include those effects that are undetectable, not measurable, or so minor that they cannot be meaningfully evaluated. Insignificant is the appropriate effect conclusion when plausible effects are going to happen, but will not rise to the level of constituting an adverse effect. That means the ESA-listed species may be expected to be affected, but not harmed or harassed.

Discountable effects are those that are extremely unlikely to occur. For an effect to be discountable, there must be a plausible adverse effect (i.e., a credible effect that could result from the action and that would be an adverse effect if it did impact a listed species), but it is very unlikely to occur.

In Greece, at least four species of sturgeon have been recorded there historically and are considered native: Stellate sturgeon (*Acipenser stellatus*), beluga sturgeon (*Huso huso*), Adriatic sturgeon (*Acipenser naccarii*), and European sturgeon (*Acipenser sturio*) (Paschos et al. 2008). Of these, only European and Adriatic sturgeon are listed under the ESA.

Although historically found in European waters from the North Sea, the English Channel, the Mediterranean and the Black Sea, European sturgeon have been mostly extirpated from their original range (Meadows 2013). European sturgeon are believed to still occur in two locations: a spawning population in southwestern France (Lepage et al. 2000), and the Rioni River system in Georgia (Kolman 2011; Meadows 2013).

The historical range of Adriatic sturgeon included the Adriatic Sea, and the coastal waters of Italy and Greece, and currently thought to only occur in the Po River in north western Italy (Meadows 2013). In 2000, there was an effort to reintroduce hatchery-reared Adriatic sturgeon fry into the River Kalamas in western Greece; however, the trial was discontinued due to pollution, overfishing, and poaching (Paschos et al. 2008).

Based on the available information, it is extremely unlikely that either European sturgeon or Adriatic sturgeon would occur in the proposed action area, and thus would not be exposed to the proposed seismic activities. The only known locations of occurrence of European or Adriatic sturgeon are well outside the proposed action area of the Hellenic Trench near Crete and the waters surrounding Santorini. We therefore conclude that the effects of the proposed action to

European and Adriatic sturgeon are discountable, and will not be considered further in this opinion.

Scalloped hammerhead shark (*Sphyrna lewini*) are distributed worldwide in coastal warm and tropical waters. The Eastern Atlantic DPS was listed as endangered, and this DPS includes scalloped hammerheads originating from the Mediterranean Sea (Miller et al. 2013). However, after discussing the proposed action with a subject matter expert and reviewing the available literature, we found that the easternmost recorded occurrence of scalloped hammerhead in the Mediterranean Sea is from San Lucido, located on the western coast of Italy (Sperone et al. 2012). Because the only known occurrence of the species is well outside the proposed action area, scalloped hammerheads are not likely to be exposed to the effects of the proposed action. We therefore conclude that the effects of the proposed action to scalloped hammerhead shark are discountable, and will not be considered further in this opinion.

Gray whales (*Eschrichitus robustus*) currently reside in the North Pacific Ocean, and are divided into two populations under the ESA: Eastern North Pacific and Western North Pacific. The Western North Pacific population is listed as endangered under the ESA. The Eastern North Pacific population was delisted from the ESA in 1994 (59 FR 31094). Gray whales are thought to have existed in the Atlantic Ocean until the 17th century but were extirpated from the region by whalers (Bryant 1995; Lindquist 2000).

In May 2010, a gray whale was sighted in the Mediterranean Sea off of Israel, and then the same individual was seen in Spanish Mediterranean waters about three weeks later, with both sightings confirmed by photo-identification (Scheinin et al. 2011b). This occurrence of a gray whale in the Mediterranean was exceptional, and prompted much speculation as to its origin. After considering relative sizes of the two extant populations, migration routes due to sea ice passages, and respective feeding areas, Scheinin et al. (2011b) postulated that it was most likely that gray whale sighted in the Mediterranean was a vagrant from the Eastern North Pacific population. To support this hypothesis, Scheinin et al. (2011) presented several points. The known summering grounds of the Eastern North Pacific gray whale population are further north in the Arctic Sea than the Western North Pacific population, which feeds in the summer in the Okhotsk Sea. A vagrant individual from the Eastern North Pacific population would be more able to find a passage in the Arctic sea ice. Furthermore, the relatively smaller Western North Pacific population (~125 individuals) has been more thoroughly catalogued through photo-identification than the larger (18,000 individuals) Eastern North Pacific population. Although photographed, the gray whale sighted in the Mediterranean was not able to be individually identified through comparison to photo catalogues, leading Scheinin et al. (2011) to believe that the Mediterranean individual came from the Eastern North Pacific population.

While it is possible that another vagrant gray whale could enter the proposed action area, we believe it is unlikely. The individual sighted in 2010 was the first sighting of a gray whale in the North Atlantic (or the Mediterranean) since the 1700s, and represents a rare occurrence. Even in the event that another gray whale is sighted within the proposed action area, we agree with the hypothesis put forth by Scheinin et al. (2011), that it is more likely that such an individual would be a gray whale from the delisted Eastern North Pacific population.

Because the only known occurrence of the species in the proposed action area is rare, gray whales are not likely to be exposed to the effects of the proposed action. We therefore conclude that the effects of the proposed action to the ESA-listed Western North Pacific gray whale DPS are discountable, and will not be considered further in this opinion.

There is no designated critical habitat found in the proposed action area, and thus, none will be affected by the proposed action. Designated critical habitat will not be considered further in this opinion.

4.2 ESA-listed Species and Critical Habitat Likely to be Adversely Affected

This opinion examines the status of each species that would be affected by the proposed action. The status is determined by the level of risk that the ESA-listed species face, based on parameters considered in documents such as recovery plans, status reviews, and listing decisions. The species status section helps to inform the description of the species' current "reproduction, numbers, or distribution" as described in 50 CFR 402.02. More detailed information on the status and trends of these ESA-listed species, and their biology and ecology can be found in the listing regulations and critical habitat designations published in the Federal Register, status reviews, recovery plans, and on these NMFS Web sites: http://www.nmfs.noaa.gov/pr/species/index.htm.

The opinion also examines the condition of critical habitat throughout the designated area, evaluates the conservation value of the various watersheds and coastal and marine environments that make up the designated area, and discusses the current function of the essential physical and biological features that help to form that conservation value.

One factor affecting the rangewide status of fin, humpback and sperm whales, Mediterranean monk seals, loggerhead, green and leatherback sea turtles, and aquatic habitat at large is climate change. Climate change will be discussed in each following section throughout the status of the species.

4.2.1 Sei Whale

4.2.1.1 Population designations

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

North Atlantic. In the western North Atlantic, a major portion of the sei whale population occurs in northern waters, potentially including the Scotian Shelf, along Labrador and Nova Scotia, south into the U.S. EEZ, including the Gulf of Maine and Georges Bank (Mitchell and Chapman 1977; Waring et al. 2004). These whales summer in northern areas before migrating south to waters along Florida, in the Gulf of Mexico, and the northern Caribbean Sea (Gambell

1985b; Mead 1977). Sei whales may range as far south as North Carolina. In the U.S. EEZ, the greatest abundance occurs during spring, with most sightings on the eastern edge of Georges Bank, in the Northeast Channel, and in Hydrographer Canyon (CETAP 1982). 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).

4.2.1.2 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. 1999a). Sei whales are often associated with deeper waters and areas along continental shelf edges (Hain et al. 1985b). This general offshore pattern is disrupted during occasional incursions into shallower inshore waters (Waring et al. 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. 1999a). When on feeding grounds, larger groupings have been observed (Gambell 1985b).

4.2.1.3 Reproduction

Very little is known regarding sei whale reproduction. Reproductive activities for sei whales occur primarily in winter. Gestation is about 12.7 months, calves are weaned at 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).

4.2.1.4 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).

4.2.1.5 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).

4.2.1.6 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 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.

North Atlantic. No information on sei whale abundance exists prior to commercial whaling (Perry et al. 1999a). 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 1982). These two estimates are more than 30 years out of date and likely do not reflect the current true abundance; in addition, the CETAP estimate has a high degree of uncertainty and is considered statistically unreliable (Perry et al. 1999a; 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.

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).

4.2.1.7 Natural threats

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

4.2.1.8 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.

Sei whales are known to accumulate DDT, DDE, and PCBs (Borrell 1993; Borrell and Aguilar 1987; Henry and Best 1983). Males carry larger burdens than females, as gestation and lactation transfer these toxins from mother to offspring.

4.2.2 Fin Whale

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

4.2.2.1 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.

4.2.2.2 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 (Mediterranean, 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).

4.2.2.3 Distribution

Fins whales are considered to be the most common mysticete species in the Mediterranean (Notarbartolo-Di-Sciara et al. 2003). Fin whales are 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. 2009a), 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. 2009a). 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).

The presence of fin whales in the Ionian and Aegean Basins (i.e., the waters surrounding Greece) is unknown, as there is a scarcity a data and published reports focusing on fin whales in those regions (Notarbartolo-Di-Sciara et al. 2003). Fin whales are considered relatively rare in Greek seas, with the majority of sightings occurring in the north Ionian Sea, and recordes of occasional strandings (5 since 1991) (Frantzis 2009). These confirmed sightings of fin whales in Greek waters are relatively fewer when compared to the Ligurian Sea (Frantzis et al. 2004a; Politi et al. 1994).

4.2.2.4 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).

4.2.2.5 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. 1992a). 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. 1999a). 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).

4.2.2.6 Habitat and 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). In the Mediterranean, fin whales have been sighted mostly in deep (>1,000-2,000 m) offshore waters, but have also been found over the continental shelf (Notarbartolo-Di-Sciara et al. 2003). The variations in their movements is likely linked to prey availability and the interannual variability in concentrations of krill, which tend to occur in locations which possess the oceanographic conditions that best support them (Aissi et al. 2008; Notarbartolo-Di-Sciara et al. 2003). Fin whales appear to show a preference for regions with high topographic variability, extensive mixing, and enhanced primary production (Hain et al. 1992b).

4.2.2.7 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. 2009a; Nasu 1974)).

4.2.2.8 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. 1999a). 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).

4.2.2.9 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 1998). 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 1997; Payne and Webb 1971). During the breeding season, fin whales produce pulses in a regular repeating pattern, which have been proposed to be mating displays similar to those of humpbacks (Croll et al. 2002). These vocal bouts last for a day or longer (Tyack 1999). The 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).

4.2.2.10 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 7**). 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).

The current population of adult fin whales in the Mediterranean is thought to be 5,000 (Otero and Conigliaro 2012); previous estimates have been from 900 to around 3,500 individuals (Forcada et al. 1995; Notarbartolo-Di-Sciara et al. 2003). The current population trend for fin whales in the Mediterranean is unknown (Frantzis 2009).

Table 7. 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. 2012)(NMFS 2008; Waring et al. 2012)
Northeastern U.S. Atlantic cont'l shelf	~~	~~	2,200-5,000	~~	(Hain et al. 1992a; 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)

*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).

4.2.2.11 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. 1999a).

4.2.2.12 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. 1997a; Gauthier et al. 1997b). Contaminant levels increase steadily with age until sexual maturity, at which time levels begin to drop in females and continue to increase in males (Aguilar and Borrell 1988).

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

4.2.2.13 Critical Habitat

NMFS has not designated critical habitat for fin whales.

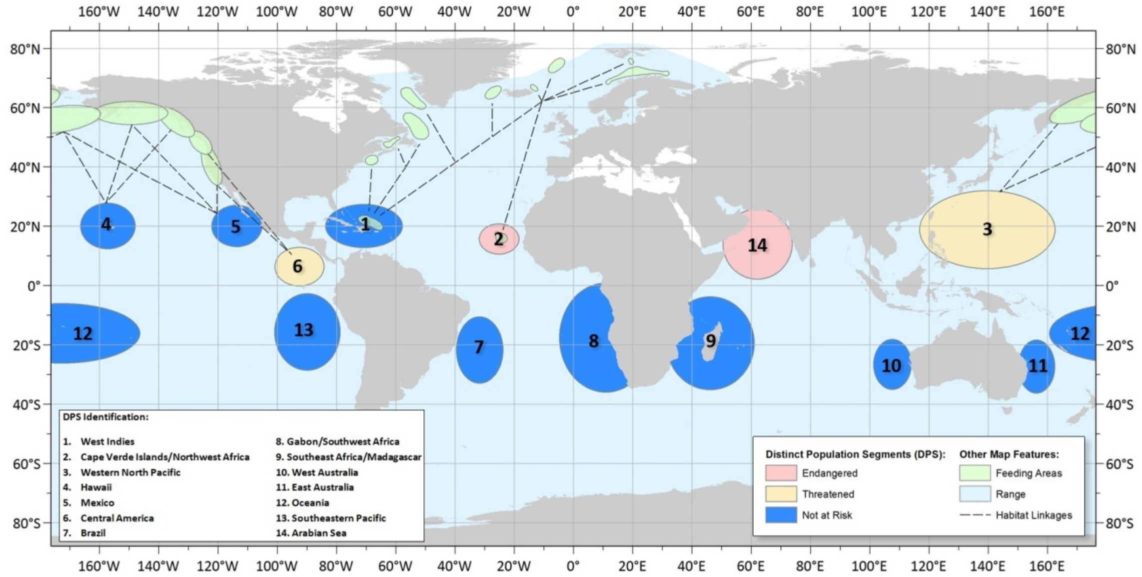
4.2.3 Humpback Whale

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 1985b).

4.2.3.1 Population Structure

Populations have been relatively well defined for humpback whales, and currently include the North Atlantic, North Pacific, Arabian Sea, and Southern Hemisphere. NMFS has proposed to revise the ESA listing for the humpback whale to identify 14 distinct population segments, with two as threatened, two as endangered, and the remaining 10 as not warranted for listing (Figure 5) (80 FR 22304).

Figure 5. Worldwide distribution of the 14 identified humpback whale distinct population segments.



Humpbacks are occasionally found in the Mediterranean Sea, but there is no evidence to support that the region supports a large humpback whale presence presently or historically (Bettridge 2015; Frantzis et al. 2004b). The population origin of these individuals sited in the Mediterranean cannot be known until further study identifies them genetically (Frantzis et al. 2004b). However, based on what is currently known about the humpback whale population structure, we can suppose that it is likely that individuals found in the Mediterranean Sea originate from the North Atlantic population (as opposed to being from the North Pacific, Arabian Sea, or Southern Hemisphere populations). It is possible that humpbacks found in the Mediterranean Sea are of the Cape Verde Islands/Northwest Africa DPS based on the proximity of that DPS to the Mediterranean Sea (Figure 5); but without genetic testing or photo-identification, it is not possible to draw any conclusions. The following section will focus on the North Atlantic humpback whale population.

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

Increasing range and occurrence in the Mediterranean Sea coincides with population growth and may represent reclaimed habitat from pre-commercial whaling (Frantzis et al. 2004c; 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. 2003c). 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.

4.2.3.2 Reproduction

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

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

4.2.3.3 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).

4.2.3.4 Feeding

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

Average group size near Kodiak Island is 2-4 individuals, although larger groups are seen near Shuyak and Sitkalidak islands and groups of 20 or more have been documented (Wynne et al. 2005).

4.2.3.5 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). 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 1986a; 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). However, humpbacks tend to be less vocal in northern feeding areas than in southern breeding areas (Richardson et al. 1995c).

4.2.3.6 Status and Trends

Humpback whales were originally listed as endangered rangewide in 1970 (35 FR 18319), and this status remains under the ESA. There are DPSs currently proposed for listing under the ESA; see Section 4.2.3.1. Winn and Reichley (1985b) argued that the global humpback whale population consisted of at least 150,000 whales in the early 1900s, mostly in the Southern Ocean. In 1987, the global population of humpback whales was estimated at about 10,000 (NMFS 1987). Although this estimate is outdated, it appears that humpback whale numbers are increasing. Table 8 provides estimates of historic and current abundance for ocean regions.

North Atlantic. The best available estimate of North Atlantic abundance comes from 1992-1993 mark-recapture data, which generated an estimate of 11,570 humpback whales (Stevick et al. 2003a). Historical estimates have ranged from 40,000-250,000 (Smith and G.Pike 2009). Smith and Reeves (2010) estimated that roughly 31,000 individuals were removed from the North Atlantic due to whaling since the 1600s. Estimates of animals on Caribbean breeding grounds exceed 2,000 individuals (Balcomb III and Nichols 1982). Several researchers report an increasing trend in abundance for the North Atlantic population, which is supported by increased sightings within the Gulf of Maine feeding aggregation (Barlow 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. (2009) 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).

Table 8. Summary of past and present humpback whale abundance.

Region	Population, stock, or study area	Pre-exploitation estimate	95% C.I.	Current estimate	95% C.I.	Source
Global	--	1,000,000	--	--	--	(Roman and Palumbi 2003)
North Atlantic	Basinwide	240,000	156,000-401,000*	11,570	10,005-13,135*	(Roman and Palumbi 2003) (Stevick et al. 2001) *in* (Waring et al. 2004)
	Basinwide - Females	--	--	2,804	1,776-4,463	(Palsbøll et al. 1997)
	Basinwide - Males	--	--	4,894	3,374-7,123	(Palsbøll et al. 1997)
	Western North Atlantic from Davis Strait, Iceland to the West Indies	>4,685*	--	--	--	*circa 1865; (Mitchell and Reeves 1983)
	NMFS - Gulf of Maine stock	--	--	845	CV=0.55	(NMFS 2008)
	NMFS - Gulf of Maine stock, including a portion of Scotian Shelf	--	--	902	177-1,627*	(Clapham et al. 2003)
	Northeast Atlantic - Barents and Norwegian Seas	--	--	889	331-1,447*	(Øien 2001) *in* (Waring et al. 2004)
North Pacific	Basinwide	15,000	--	6,000-8,000	--	(Calambokidis et al. 1997)
	NMFS - Western North Pacific stock	--	--	394	329-459*	(Angliss and Allen 2007)
	NMFS - Central North Pacific stock	--	--	4,005	3,259-4,751*	(Angliss and Allen 2007)
	NMFS - Eastern North Pacific stock	--	--	1,391	1,331-1,451*	(Carretta et al. 2008)
Indian Ocean	Arabian Sea	--	--	56	35-255	Minton et al. (Minton et al. 2003) *in* (Bannister 2005)
Southern Hemisphere	Basinwide	100,000	--	19,851	--	(Gambell 1976; IWC 1996)
	South of 60°S	--	--	4,660	2,897-6,423	(IWC 1996)

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

4.2.3.7 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. 1999a). 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).

4.2.3.8 Anthropogenic Threats

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

Humpback whales are also killed or injured during interactions with commercial fishing gear. Like fin whales, 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. 2007d). Of these, 95 entangled humpback whales were confirmed, with 11 whales sustaining injuries and nine dying of their wounds. Between 30 and 40% of humpback whales in the Arabian Sea show scarring from entanglements, with fishing effort on the rise (Baldwin et al. 2010).

More humpback whales are killed in collisions with ships than any other whale species except fin whales (Jensen and Silber 2003). Of 123 humpback whales that stranded along the Atlantic coast of the U.S. between 1975 and 1996, 10 (8.1%) showed evidence of collisions with ships (Laist et al. 2001). Between 1999 and 2005, there were 18 reports of humpback whales being struck by vessels along the Atlantic coast of the U.S. and the Maritime Provinces of Canada (Cole et al. 2005c; Nelson et al. 2007d). Of these reports, 13 were confirmed as ship strikes and in seven cases, ship strike was determined to be the cause of death. In the Bay of Fundy, recommendations for slower vessel speeds to avoid right whale ship strike appear to be largely ignored (Vanderlaan et al. 2008). 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%. The first estimate of population-level effects of entanglement were recently produced, with over 12% of the Gulf of Maine population of humpbacks acquiring new scars from entanglement interactions annually (Mattila and Rowles 2010).

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

4.2.3.9 Critical habitat.

NMFS has not designated critical habitat for humpback whales.

4.2.4 Sperm Whale

Sperm whales are distributed in all of the world's oceans, from equatorial to polar waters, and are highly migratory. Mature males range between 70º N in the North Atlantic and 70º S in the Southern Ocean (Perry et al. 1999a; 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.

4.2.4.1 Stock Designations

There is no clear understanding of the global population structure of sperm whales (Dufault et al. 1999). Recent ocean-wide genetic studies indicate low, but statistically significant, genetic diversity and no clear geographic structure, but strong differentiation between social groups (Lyrholm and Gyllensten 1998; Lyrholm et al. 1996; Lyrholm et al. 1999). The IWC currently recognizes four sperm whale stocks: North Atlantic, North Pacific, northern Indian Ocean, and Southern Hemisphere (Dufault et al. 1999; Reeves and Whitehead 1997). 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. 1999a; 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). Genetic studies have indicated that Mediterranean sperm whales are genetically distinct from the Atlantic stocks (Drouot et al. 2004; Engelhaupt et al. 2009), but there are doubts as to whether this information provides enough support for a within-region population structure (Reeves and Sciara 2006).

Sperm whales are considered regular inhabitants of the Aegean Sea, particularly around the Hellenic Trench (Frantzis 2009; Reeves and Sciara 2006). In the Mediterranean, sperm whales are found from the Alboran Sea to the Levant Basin, primarily over steep slope and deep offshore waters. Sperm whales are rarely sighted in the Sicilian Channel, and are vagrants to the northern Adriatic and Aegean seas (Notarbartolo di Sciara and Demma 1997). In Italian seas, sperm whales are more frequently associated with the continental slope off western Liguria, western Sardinia, northern and eastern Sicily, and both coasts of Calabria.

4.2.4.2 Movement

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

4.2.4.3 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). When they are found relatively close to shore, sperm whales are usually associated with sharp increases in topography where upwelling occurs and biological production is high, implying the presence of a good food supply (Clarke 1956). Such areas include oceanic islands and along the outer continental shelf.

Sperm whales are frequently found in locations of high productivity due to upwelling or steep underwater topography, such as continental slopes, seamounts, or canyon features (Jaquet and Whitehead 1996; Jaquet et al. 1996). Cold-core eddy features are also attractive to sperm whales in the Gulf of Mexico, likely because of the large numbers of squid that are drawn to the high concentrations of plankton associated with these features (Biggs et al. 2000; Davis et al. 2000a; Davis et al. 2000b; Davis et al. 2000c; Davis et al. 2002; Wormuth et al. 2000). Surface waters with sharp horizontal thermal gradients, such as along the Gulf Stream in the Atlantic, may also be temporary feeding areas for sperm whales (Griffin 1999; Jaquet et al. 1996; Waring et al. 1993). Sperm whales over George's Bank were associated with surface temperatures of 23.2-24.9° C (Waring et al. 2003). Water depth also appears to play a role in sperm whale presence; 188 sightings of sperm whales along the Hellenic Trench occurred at a mean depth of 1,235 m (510-2,933 m) and a mean distance of 7.9 km from shore (Frantzis 2009). A subsequent study in the same area found a peak in sperm whale distribution along the 1,000 m contour (Frantzis et al. 2014).

4.2.4.4 Reproduction

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

Sperm whale age distribution is unknown, but sperm whales are believed to live at least 60 years (Rice 1978). Estimated annual mortality rates of sperm whales are thought to vary by age, but previous estimates of mortality rate for juveniles and adults are now considered unreliable (IWC 1980). In addition to anthropogenic threats, there is evidence that sperm whale age classes are subject to predation by killer whales (Arnbom et al. 1987; Pitman et al. 2001).

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

During their breeding prime and old age, male sperm whales are essentially solitary (Christal and Whitehead 1997).

4.2.4.5 Diving

Sperm whales are probably the deepest and longest diving mammalian species, with dives to 3 km down and durations in excess of 2 hours (Clarke 1976; Watkins et al. 1993; Watkins et al. 1985). However, dives are generally shorter (25- 45 min) and shallower (400-1,000 m). Dives are separated by 8-11 min rests at the surface (Gordon 1987; Jochens et al. 2006; Papastavrou et al. 1989; Watwood et al. 2006; Würsig et al. 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.

4.2.4.6 Feeding

Sperm whales appear to feed regularly throughout the year (NMFS 2006). It is estimated they consume about 3-3.5% of their body weight daily (Lockyer 1981). They seem to forage mainly on or near the bottom, often ingesting stones, sand, sponges, and other non-food items (Rice 1989a). A large proportion of a sperm whale’s diet consists of low-fat, ammoniacal, or luminescent squids (Clarke 1996; Clarke 1980b; Jaquet and Gendron 2002; 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).

4.2.4.7 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).

4.2.4.8 Status and Trends

Sperm whales were originally listed as endangered in 1970 (35 FR 18319), and this status remained with the inception of the ESA in 1973. Although population structure of sperm whales is unknown, several studies and estimates of abundance are available. Table 9 contains historic and current estimates of sperm whales by region. Sperm whale populations probably are undergoing the dynamics of small population sizes, which is a threat in and of itself. In particular, the loss of sperm whales to directed Soviet whaling likely inhibits recovery due to the loss of adult females and their calves, leaving sizeable gaps in demographic and age structuring (Whitehead 2003).

There is no definitive population estimate for sperm whales in the Mediterranean, but their numbers are thought to be in the low hundreds (Otero and Conigliaro 2012). Photo-id studies in the area of the Hellenic Trench indicate a local population size of 200-250 individuals (Frantzis et al. 2014).

Table 9. Summary of past and present sperm whale abundance.

Region	Population, stock, or study area	Pre-exploitation estimate	95% C.I.	Current estimate	95% C.I.	Source
Global	--	--	--	900,000	--	(Würsig et al. 2000)
	--	1,110,000	672,000-1,512,000	360,000	105,984-614,016*	(Whitehead 2002)
North Atlantic	Basinwide	224,800	--	22,000	--	(Gosho et al. 1984; Würsig et al. 2000)
	Northeast Atlantic, Faroes-Iceland, and U.S. East Coast (combined)	--	--	13,190	--	(Whitehead 2002)
	NMFS - North Atlantic stock (Western North Atlantic)	--	--	4,804	1,226-8,382*	(NMFS 2008)
	Eastern North Atlantic - Iceland	--	--	1,234	823-1,645*	(Gunnlaugsson and Sigurjónsson 1990)
	Eastern North Atlantic - Faroe Islands	--	--	308	79-537*	(Gunnlaugsson and Sigurjónsson 1990)
	Eastern North Atlantic - Norwegian Sea	--	--	5,231	2,053-8,409*	(Christensen et al. 1992b)
	Eastern North Atlantic - Northern Norway to Spitsbergen	--	--	2,548	1,200-3,896*	(Øien 1990)
Gulf of Mexico	NMFS - Gulf of Mexico stock	--	--	1,665	CV=0.2	(NMFS 2008)
	Northern Gulf of Mexico - off the Mississippi River Delta between 86° and 91°W	--	--	398	253-607	(Jochens et al. 2006)
	North-central and Northwestern Gulf of Mexico	--	--	87	52-146	(Mullin et al. 2004)
North Pacific	Basinwide	620,400	--	472,100	--	(Gosho et al. 1984)
				930,000	--	(Rice 1989a)
	Eastern Tropical Pacific	--	--	26,053	13,797-38,309*	(Whitehead 2003)
	Off Costa Rica	--	--	1,360	823-2,248*	(Gerrodette and Palacios 1996)
	Off Central America north of Costa Rica	--	--	333	125-890*	(Gerrodette and Palacios 1996)
	Eastern Temperate North Pacific	--	--	26,300	0-68,054*	(Barlow and Taylor 2005)
				32,100	9,450-54,750*	(Barlow and Taylor 2005)
	NMFS - North Pacific stock	--	--	--	--	(Angliss and Allen 2007)
	NMFS - California/Oregon/ Washington stock	--	--	2,853	CV=0.25*	(Carretta et al. 2008)
	NMFS - Hawaii stock	--	--	7,082	2,918-11,246*	(Carretta et al. 2008)
Southern Hemisphere	Basinwide	547,600	--	299,400	--	(Gosho et al. 1984; IWC 1988; Perry et al. 1999a)
	South of 60°S	--	--	14,000	8,786-19,214*	(Butterworth et al. 1995) as cited in (Perry et al. 1999a)
	South of 30°S	--	--	128,000	17,613-238,387*	(Butterworth et al. 1995) as cited in (Perry et al. 1999a)

*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).

4.2.4.9 Natural Threats

Sperm whales are 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).

4.2.4.10 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). Still, 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 2004b). Whale-watching vessels are known to influence sperm whale behavior (Richter et al. 2006). An individual was caught and released from gillnetting, although injured, on Georges Bank during 1990. A second individual was freed, but injured, from gillnetting on George’s Bank in 1995. In 1994, a sperm whale was disentangled from gillnet along the coast of Maine. In August 1993, a dead sperm whale, with longline gear wound tightly around the jaw, was found floating ~32 km off Maine.

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

4.2.4.11 Critical Habitat

NMFS has not designated critical habitat for sperm whales.

4.2.5 Mediterranean Monk Seal

4.2.5.1 Populations

Greece hosts the largest Mediterranean monk seal population (Adamantopoulou et al. 2000; Adamantopoulou et al. 2011), with small, isolated groups or individuals in other locations along with other concentrations along Mauritania on the Atlantic coast of Africa (Sergeant et al. 1978). Three islands off the coast of Madeira, Portugal, known as the Desertas Islands, host an additional, isolated collection of a few monk seals (CMS 2005).

4.2.5.2 Distribution

Mediterranean monk seals are generally found throughout islands and marine waters of Greece (Adamantopoulou et al. 1998; Adamantopoulou et al. 1999; MOm 2014a) (Figure 6). Once stretching from the Azores into and throughout the Mediterranean Sea, the Black Sea, and along the West African coast south to Gambia, Mediterranean monk seal distribution is now limited to isolated pockets or rare sightings. Populations in the Azores and the Black Sea are now considered extirpated. In portions of this former range sightings are rare and have occurred in places such as Croatia, Israel, Sicily, the Cape Verde and Canary Islands, Morocco, Senegal, and Gambia (Anonymous 2001; Bareham and Furreddu 1975; Brito et al. 2006; Bundone et al. 2013; CMS 2005; Gomercic et al. 2011; Kirac and Savas 1996; Marchessaux 1989; Mo 2011; Mo et al. 2011; MOm 2014a; Scheinin et al. 2011a; Sergeant et al. 1978) (Figure 6). Monk seals appear to have once been abundant along the Sardinian coast (De Waele et al. 2009). A “large colony” was rumored to exist on the coast of Karabiga (opening to the Black Sea), Turkey roughly 30 years ago (Inanmaz et al. 2014).

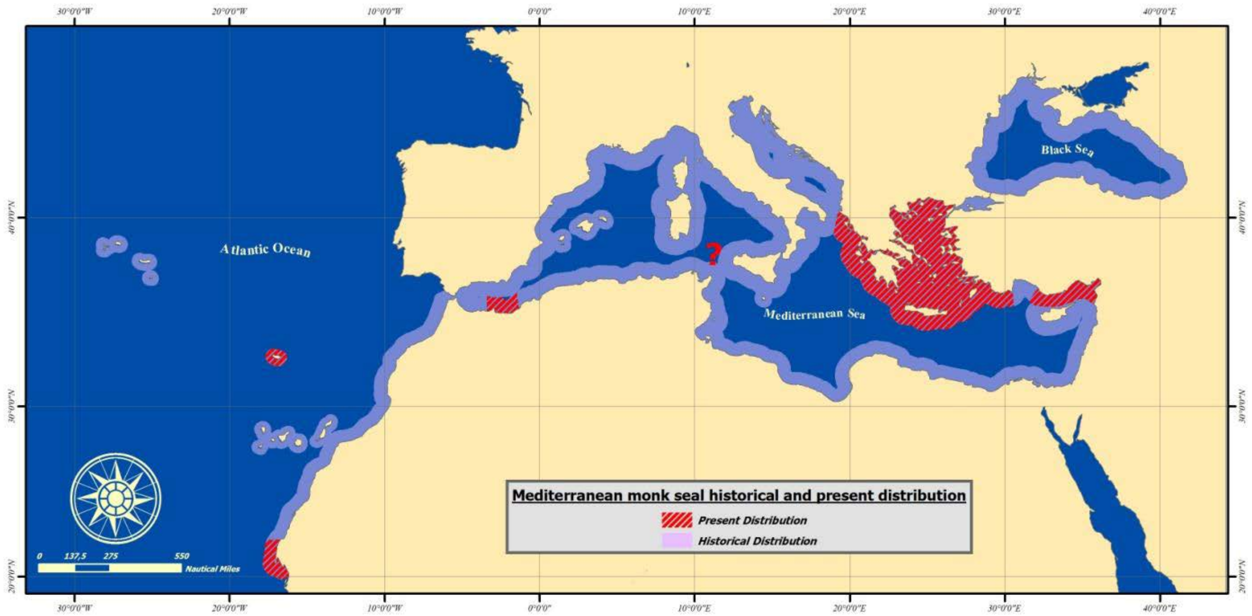


Figure 6. Historical and current Mediterranean monk seal distribution. Figure taken from MOm (2014a).

The proposed action takes place in two locations: a single trackline through the Hellenic Trench (northwest of Crete) and approximately 1,258km of tracklines around Santorini and south of the Cyclades island group. Mediterranean monk seals have been observed or studied on numerous islands throughout Greek waters.

In the 1970s, the Aegean Sea was the center of the Mediterranean population (Sergeant et al. 1978). Marchessaux and Duguy (1977) provided one of the earlier comprehensive descriptions of Mediterranean monk seals in Greek waters. This review included records of monk seals on Greek islands included in the Cyclades group, (e.g., Mykonos, Syros, Amorgos, Santorini, Astipalaia) and other island groups in the region, like the Northern Sporades, Ionian Islands, Crete, and the Dodecanedes. Using these observations, Marchessaux and Duguy generated population estimates for Mediterranean monk seals in Greece (Table 10). Figure 7 provides a map of the region to illustrate the 1977 population estimates, with the general locations of the current proposed actions.

Table 10 Mediterranean monk seal population estimates for Greek waters in 1977 Marchessaux and Duguy 1977.

Island Group/Region	1977 Population Estimate (Individuals; min-max)
Continental Greece	20-30
Sporades	30-50
Ionian Islands	30-40
Cyclades	70-90
Dodecanedes	80-100
Crete and neighboring islands	30-50

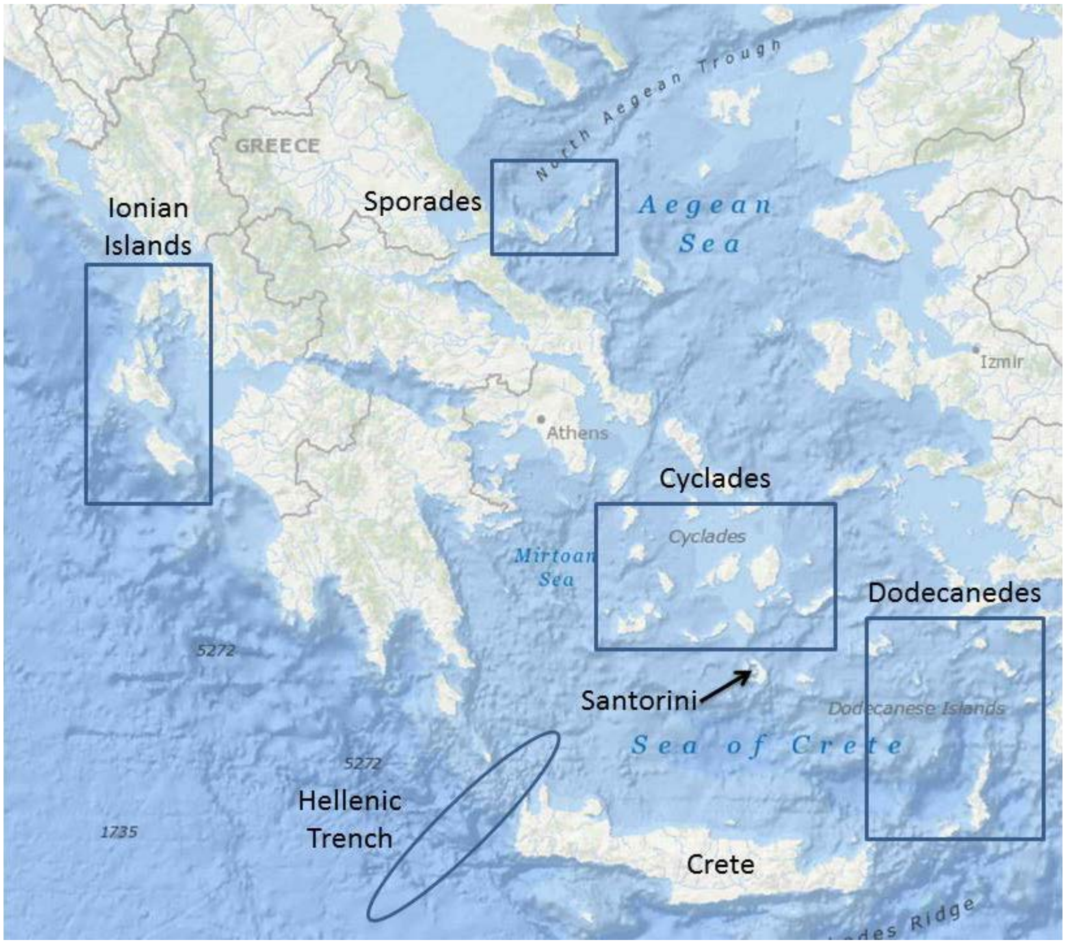


Figure 7 Map of Greece and the general locations of major island groups with historical population estimates for Mediterranean monk seals (Marchesseaux and Duguy 1977).

Santorini and the Hellenic Trench subduction zone (the general sites of the proposed action) are identified.

Historically, Mediterranean monk seal occurrence in the Cyclades Islands is poorly known (Marchessaux and Duguy 1977). On Syros pupping is known to occur during spring and roughly 15 individuals were estimated here in 1976. A group of 30 monk seals were estimated on Amorgos Island in the 1970s, including pups. On Santorini and neighboring islands, 15 monk seals were estimated in the 1970s, including eight along the northern and northeastern edge of the island, including pups. One adult male monk seal washed ashore dead here in 1990 (Menchero et al. 1994). Monk seals were not observed along Anafi Island historically. Small numbers (3-5 individuals) have been reported from Astypalaia Island over several decades (Marchessaux and Duguy 1977).

On the Island of Patmos (of the Dodecanese Islands), 4-5 seals were seen annually on a regular basis, but at the time, local fishermen reported that these numbers were declining into the 1970s (Marchessaux and Duguy 1977). Along the Turkish coast on the island of Kos, a colony of about 20 individuals was studied in the 1970s (Marchessaux and Duguy 1977; Sergeant et al. 1978). A few individuals were seen on Nisiros Island, just south of here in 1972 and 1973 (Marchessaux and Duguy 1977). Another few individuals were seen around the nearby island of Symi (Marchessaux and Duguy 1977). Roughly 100 miles east of the action area, the island of Rhodes along the Turkish coast was thought to host roughly 40 monk seals in the 1800s (Marchessaux and Duguy 1977).

Although monk seals were historically numerous on Crete and its surrounding islands, their numbers were severely reduced by the 1970s (Marchessaux and Duguy 1977).

In recent years, there have been continuing efforts to study Mediterranean monk seals in Greek waters, which remain widely distributed in Greece (MOm 2014a) (Figure 8). There have been sightings of monk seals in places where they were not previously observed. The Cyclades Islands group has been the focus of much study. The north side of Anafi Island is now a known location of pupping and resting, where an estimated minimum of 17 individuals reside (MOm 2014a). Pups have been observed here in November and December (MOm 2014a). In the Kimolos Island complex, 24 monk seal shelters have been identified, hosting 25-40 individuals and roughly six births annually (Adamantopoulou et al. 2000; Karamanlidis et al. 2010).

In the Dodecandese Karpathos Islands, 14 shelters are known supporting 10-20 individuals and roughly two to three births per year (Adamantopoulou et al. 2000). In the Dodecandese Fournoi Islands, 24 shelters have been found, hosting 5-10 animals and about one birth per year (Adamantopoulou et al. 2000).

The Ionian islands of Ithaca and Kefallinia have multiple haul-out sites (Hiby and Jeffery 1987). At least 25 individuals occur on the island of Gyaros, including use of caves and open beaches

(Karamanlidis et al. 2013). This is also thought to be the only case of a colony structure in the eastern Mediterranean Sea that was historically normal throughout the species range (Karamanlidis et al. 2013).

Individuals from Greek waters have also been sighted more broadly, such as along Israel, Libya, the Marmara sea in Turkey and Croatia (Adamantopoulou et al. 2011). The first record (bycatch) of a Mediterranean monk seal along the Libyan coast (a former host to monk seal colonies) was recently made for the first time in 40 years and was genetically consistent with individuals from Crete and the Aegean Sea (Alfaghi et al. 2013).

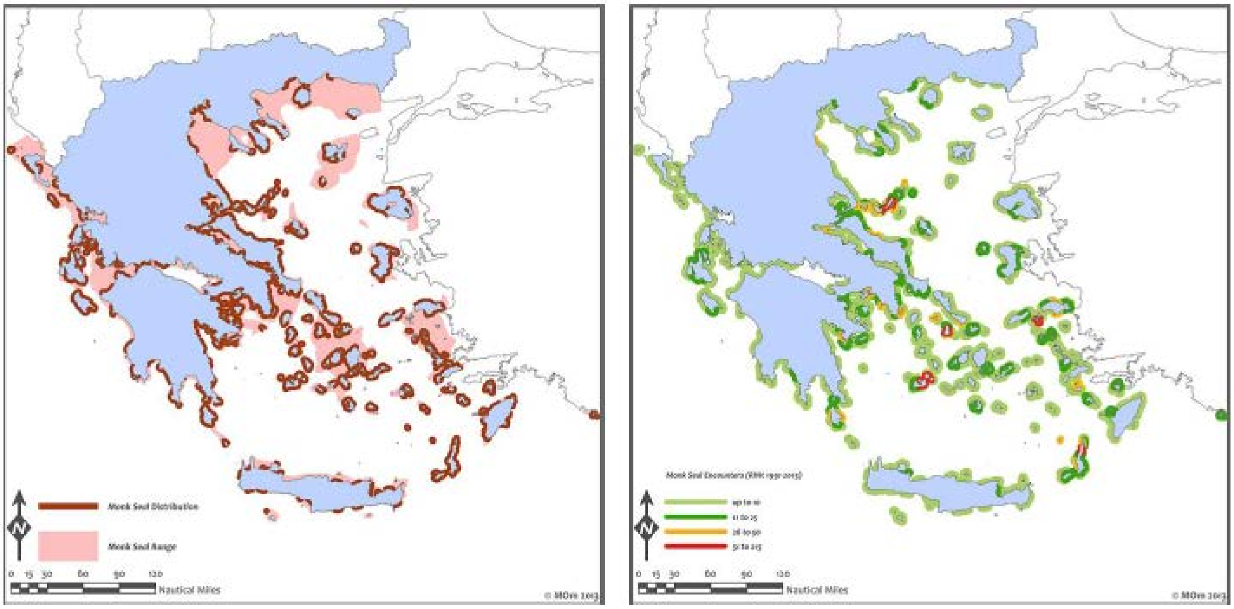


Figure 8. Distribution and sightings of Mediterranean monk seals in Greek waters. Figure taken from MOm 2014.

4.2.4.3 Habitat

Mediterranean monk seals occupy shallow coastal waters, but (particularly juveniles) can make extended trips through deeper waters (CMS 2005; MOm 2014a). Prior to heavy human exploitation, Mediterranean monk seals were commonly found on sandy beaches and atolls (CMS 2005; Gonzalez et al. 2002; Sergeant et al. 1978). However, these individuals were more easily harvested and remaining individuals are generally found in locations dominated by caves and grottos accessible by sea, or on secluded beaches (CMS 2005; Marchessaux and Duguy 1977). Up to 100 individuals have been observed to simultaneously haul-out in such locations (CMS 2005). The Cape Blanco region is a one of high marine productivity, likely a factor in supporting the existing colony there (CMS 2005). Here, males occupy and apparently defend marine territory near cliff faces where mating occurs while adult females and young occur in certain cave systems at these locations (CMS 2005).

Habitat in Greece tends to be rocky, isolated locations (Azzolin et al. 2014; Sergeant et al. 1978). Unlike most other seal species, Mediterranean monk seals are known to haul-out in grottos or caves frequently accessible only by underwater entrances, possibly as a mechanism to offset human disturbance (Bareham and Furreddu 1975; Bayed et al. 2005; CMS 2005; Dendrinos et al. 2007b). Movement into and out of these locations is not clearly tied to sea or tide state, day or night, or sea/air temperature in some cases (Bareham and Furreddu 1975; Dendrinos et al. 2001; Marchessaux and Duguy 1977; Sergeant et al. 1978). Several caves suitable for pupping and/or resting occur in or near the action area (Dendrinos et al. 2008).

Available data from Greece indicate that Mediterranean monk seals appear to have fairly restricted home ranges (from about 100 to 1,000 km^2) based on five telemetry tagged monk seals in the eastern Mediterranean sea (Adamantopoulou et al. 2011). Home ranges of adult male seals in the Cilician Basin in southern Turkey were between 37 and 56 km (Gucu et al. 2004). A study focusing on a juvenile male in the Northern Sporades found that it exhibited a similar home range (Dendrinos et al. 2007a). Although their primary habitat seems to be nearshore shallow waters within the 200m isopleth (Dendrinos et al. 2007a), individuals are known to move over deep (>200m) oceanic waters (Adamantopoulou et al. 2011; Dendrinos et al. 2007a; Sergeant et al. 1978). Based on the typical home ranges and the bathymetry of the eastern Mediterranean Sea, the 200m isopleth is relatively close to shore (~5km), a Mediterranean monk seal could routinely travel over waters greater than 200m deep.

Based upon a few tagged individuals of the Cabo Blanco colony, lactating female Mediterranean monk seals generally dive in waters 40-60 m deep (CMS 2005), although deeper dives have been recorded. Satellite telemetry from a juvenile male that was rehabilitated and released in the eastern Mediterranean reported a mean maximum dive of 41 m deep, with a maximum depth of 123 m (Dendrinos et al. 2007a). This same individual had previously been reported to dive to a maximum of 180 m[2]. Monk seals may focus on areas shallower (2-25 m deep) than this while foraging (CMS 2005). Pups tend to remain in shallow, nearshore waters (~10 m deep) and gradually distribute further from natal caves into waters up to 40 m deep (CMS 2005; Gazo 1997a; Gazo et al. 2006). These findings generally comport with older chronicles prior to the advent of data-logging technology (Sergeant et al. 1978).

4.2.5.3 Growth and reproduction.

A single individual monitored at Cabo Blanco was reported to reach sexual maturity at 2.5 years (Gazo 2000). This is the earliest age of sexual maturity recorded for Mediterranean monk seals, with other reports placing the age at 4 years, or even 5-6, with an estimate 3-4 years for an individual aged in the eastern Mediterranean (Gazo 2000; Marchessaux 1988). In their stochastic

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[2] http://www.monachus-guardian.org/mguard14/1414mednew.htm#Orphanedseal

population viability analysis for the eastern Atlantic Mediterranean monk seal, the authors assumed that 4 years old was the age of first reproduction (CMS 2005).

Pupping does occur year-round for the colony at Cabo Blanco, but pupping is less frequent from October through February (Aguilar et al. 1995; Cedenilla et al. 2007; Gazo et al. 1999b; Gonzalez et al. 1994; Pastor and Aguilar 2003). The ability of the Cabo Blanco colony to pup year round in this location is likely due to the subtropical climate and the presence of a semi-permanent upwelling (Pastor and Aguilar 2003). The pupping season for the species appears to vary over a broad geographical scale, and is possibly influenced by latitudinal differences—the Cabo Blanco peninsula lies at ~20°N, while Greece is at 39-40°N (Pastor and Aguilar 2003). Pupping season for Mediterranean monk seals in Greek waters occurs from August to December, with a peak in September to October (MOm 2014).

Breeding in the action area is highest during July-December (Dendrinos et al. 1999; Sergeant et al. 1978) and newborns on the island of Patmos (in the Dodecanedes islands group) were documented for the months of September and October in the 70s (Marchessaux and Duguy 1977). Pupping was also observed in the Cyclades on Kimolos Island in mid-September 2007 (MOm 2014a).

Copulation occurs in the water outside of caves, which are aggressively defended by males (Pastor et al. 2011). Males defend the same aquatic territories year-round and across years, mating with multiple females (CMS 2005). Gestation is likely 9-10 months long (Marchessaux 1988). Adult females appear to produce a single pup on roughly an annual- to every third-year basis (CMS 2005; Gazo et al. 1999b). Pups are born inside caves and grottos, the appropriate conditions of which can be very limited (only two regular pupping caves are known from Cabo Blanco and 16 [three actually used] in the Maderas/Desertas Islands complex) (CMS 2005; Dendrinos et al. 2007b; Gazo et al. 1999b). Open-beach pupping may be expanding in the Desertas colony (Gonzalez et al. 2002; Pires and Neves 2000; Pires and Neves 2001) and may be a maternally-inherited behavior (Pires and Neves 2000; Pires and Neves 2001). Open beach pupping location exists for a few monk seals on the Greek Island of Gyaros in the central Aegean Sea (Dendrinos et al. 2008).

In the western Saharan sub-population, mothers spend roughly a third of their time nursing or bonding with pups in the first week after birth (Aguilar et al. 2007). This declined by roughly two-thirds over the remainder of the nursing period (Aguilar et al. 2007). During the first week postpartum, mothers remained with pups continuously, but started to leave pupping caves in the second week to forage (Aguilar et al. 2007; Gazo and Aguilar 2005). Pups frequently suckle from other females (Aguilar et al. 2007; Cedenilla et al. 2009), perhaps as a communal solution to long foraging trips that mothers take. Females may successfully “steal” and nurse pups from other mothers (Cedenilla et al. 2009). Sergeant et al. (1978) suggested that exposure to human presence can increase abortion rates in Mediterranean monk seals. Older pups can travel up to a couple of kilometers from the natal cave (CMS 2005). Weaning in the Cabo Blanco colony occurrs after an average of 119 days (range 103-149), almost double the lactation period of any other seal species (Aguilar et al. 2007; CMS 2005; Pastor and Aguilar 2003); a lactation period of 4-5 months has been observed for seals in the Desertas population as well CMS 2005.

4.2.5.4 Feeding.

Based on observations of two specimens in captivity, adult Mediterranean monk seals are believed to require 5% of their body weight daily in prey (CMS 2005; Sergeant et al. 1978). Monk seals forage on anchovies and pilchards, along with benthic cephalopods, fishes, and crustaceans (Boutiba and Abdelghani 1996; CMS 2005; Guclusoy 2008b; Karamanlidis et al. 2011; Pierce et al. 2011; Pierce et al. 2009; Salman et al. 2001; Sergeant et al. 1978). Based upon stomach samples from Cabo Blanco and Greece, Octopus vulgaris represented one-third to one-half of prey species identified (CMS 2005; Pierce et al. 2011). Monk seals in Greek waters have been observed predating on adult loggerhead sea turtles (Margaritoulis and Touliatou 2011). They have also been known to eat cultivated grapes, potatoes, tomatoes, and other crops planted near beaches (Boutiba and Abdelghani 1996). Monk seals have attacked aquaculture pens containing sea bream and sea bass along the Turkish Aegean coast (Guclusoy and Savas 2003a). Individual variation in prey selection is high (CMS 2005), but appears to be coastal in nature (Karamanlidis et al. 2014).

Individuals may travel for several days to foraging locations. Foraging dives for males averaged 25.5 m (maximum of 58 m) and averaged 3.5 minutes in duration (maximum of 8 minutes) (CMS 2005; Gazo 1997b). At Cabo Blanco, lactating females dove somewhat longer and deeper to an average depth of 28-38 m (maximum of 78) for an average of 5-6 minutes (CMS 2005; Gazo and Aguilar 2005). Females were gone for an average of 9.5 hours, but up to 17.4 hours to areas at least 23-39 km from caves, diving more during nighttime than daytime (Gazo and Aguilar 2005). However, fisheries interactions with monk seals have occurred 5-20 nautical miles offshore from the Saharan coast (Ridgway and Harrison 1981). In Greek waters, seals may generally stay closer to their haul-out locations, and return to their caves each evening (Marchessaux and Duguy 1977).

4.2.5.5 Vocalization and hearing.

Adult female Mediterranean monk seals vocalize in air within a range of 438-3,050 Hz, while pups call in a range of 269-1,584 Hz (Munoz et al. 2011).

4.2.5.6 Status.

Fewer than 600 individuals are thought to survive at present (Alfaghi et al. 2013; Bundone et al. 2013), and may number 400-500 individuals (Azzolin et al. 2014). Major breeding centers along the eastern Atlantic at Desertas Islands (Madeira) and Cabo Blanco are believed to host fewer than 200 individuals in genetically isolated, restricted habitats (Anonymous 2001). Both have undergone population declines, both recent and over the long-term (Anonymous 2001; Bayed and Beaubrun 1987).

Dhakla Bay, Morocco (encompassing Cabo Blanco) was reported to have 5,000 individuals hundreds of years ago (CMS 2005). Cabo Blanco likely hosted roughly 300 individuals in the 1950s, but declined to roughly 80 individuals in the 1960s and 60 or less by the 1970s (CMS 2005; Maigret 1986). From 1993-1994, 113-165 non-pup individuals were estimated at Cabo Blanco (Gonzalez et al. 1997). Just prior to 1997, roughly 300 individuals were likely present at Cabo Blanco and the sub-population was likely stable from 1993-1997 (CMS 2005). From 1994-1998, 256 individuals were identified in the first systematic surveys of the colony (CMS 2005). From 1993-1996, 44-58 pups were born annually at Cabo Blanco and the population was estimated at 317 individuals (CI: 237-447) (Forcada et al. 1999; Gazo et al. 1999b).

In 1997, a major die-off occurred here, where two-thirds of seals died and significantly altered the colony’s age structure (Cedenilla et al. 2007; CMS 2005). Only 109 seals remained (CI: 86-145) and only 77 potentially reproductive individuals (Forcada et al. 1999). The proportion of juveniles and medium-sized seals increased from 10% to 29% and 29% to 35%, respectively (CMS 2005; Forcada et al. 1999). In contrary, the proportion of adult females and males decreased from 44% to 26% and 18% to 10%, respectively (CMS 2005; Forcada et al. 1999). Not surprisingly, pup production declined from 1998-2004 to 23-29 pups born per year (CMS 2005; Gonzalez et al. 2002). However, pup survival rates increased from pre to post die-off periods, suggesting pup production may be density-dependent and capped by available habitat (caves) (Gazo et al. 1999a; Gonzalez et al. 2002). Even with this die-off, the extinction probability has not yet greatly increased for the colony, but would do so if another catastrophic event were to occur here (CMS 2005). The current population trajectory of the colony is -3.5% annually, including the major die-off (CMS 2005). Excluding this mortality event, the colony seems to be recovering (Martinez-Jauregui et al. 2012). Roughly 150 individuals were believed to live here in 2005 (CMS 2005). However, the ability to statistically detect important colony trends is poor (Forcada 2000).

Madeira (including Desertas Island), off the coast of Portugal, is thought to have once hosted about 1,600 individuals (Brito et al. 2006). This was greatly reduced after the Portuguese landed and colonized the area in 1420 (CMS 2005). By the 1970s, roughly 50 individuals are thought to have remained and continued to decline to 6-8 individuals in the 1980s. The population appears to have increased to 28 individuals by 2004 (CMS 2005). The latest estimate in 2008 is that 20-30 individuals survive here (Pires et al. 2008). Pup production for the four known reproductive females has amounted to 35 individuals from 1989-2005, and annual birth rates have gradually increased over the same period. These increases may be a result of expanded habitat, both in use of open beaches as well as additional undisturbed island locations (Pires et al. 2008). The two major Atlantic groups, Madeira and Cabo Blanco, are believed to represent roughly half of all

Mediterranean monk seals and are collectively believed to be about 3% of their former abundance (CMS 2005).

Roughly 180-300 individuals are believed to live in Greek waters, making Greece one of the last strongholds for the species (Azzolin et al. 2014; MOm 2014a), and 300-350 in the broader eastern Mediterranean (MOm 2014a). The sub-population size seems to have contracted in size since the 1970s, when 310 individuals were estimated in the region (Marchessaux and Duguy 1977). The range of the species has also contracted within the region versus historical times, in the past due to harvesting but more recently due to fisheries interactions (Marchessaux and Duguy 1977). 18 individuals were identified in the Ionian Sea (Panou et al. 1993), with the remaining large majority in the Aegean Sea and surrounding areas.

Along the Turkish coast, nine individuals have been identified in the Foça Pilot Monk Seal Conservation Area, six of which are adult females utilizing coastal caves for pupping (Guclusoy and Savas 2003b). Another four individuals were found in caves of Olympos-Beydaglari National Park, Turkey (Gucu et al. 2009) and small numbers along various regions of the coast (Ozturk 1997; Ozturk 1998; Ozturk 1999). Population viability analysis suggests the risk of extinction along the Turkish coast is declining, but risks remain due primarily to mortality of adult females, skewed sex ratio, Allee effects, and inbreeding stress (Saydam et al. 2014).

At Cabo Blanco, mortality rates are estimated at 52.1% (females) and 48.9% (males) for the first year of life and decline to 10.4% (females) and 6.7% (males) in the second and third (4.2% for females, 2.2% for males) years, respectively, and 3.4% (SD=0.3) for each year thereafter (CMS 2005; Munoz-Canas et al. 2013). Mortality was 53% through the first two months (Gazo et al. 2000). Roughly 32% of pups reached sexual maturity in the Cabo Blanco colony (CMS 2005). Sub-adult, adult male, and adult female survival rates are 81, 99, and 87%, respectively (Martinez-Jauregui et al. 2012). Maximum longevity at Cabo Blanco is poorly known and, based upon two individuals, may be 24-44 years (CMS 2005).

In the Greek subpopulation, 45 deceased individuals were aged to be up to 36 years old (Murphy et al. 2012). Most (65%) were adults, 20% juveniles, and 15% sub-adults (Murphy et al. 2012), a somewhat skewed age distribution underrepresenting sub-adults.

4.2.5.7 Natural threats.

Pup mortality is primarily due to starvation and secondarily due to disease, crushing by adults, and predation (CMS 2005; Gazo et al. 2000). Killer whales occur in the region of the Cabo Blanco colony and may predate on monk seals, but this has not been documented (CMS 2005). Heartworms have been identified in Mediterranean monk seals (Papadopoulos et al. 2010).

Very few caves suitable for pupping exist in areas where monk seals occur. Although historical records indicate that pupping on beaches, the species seems to have drastically altered its behavior and biology in response to human exploitation. As a consequence, the habitat preferred for pupping seems to have been significantly limited to certain sea-entry caves (Gonzalez et al. 2002). These caves undergo frequent collapses, which can injure or kill individuals that are present (Gonzalez et al. 1997). The 1997 die-off at Cabo Blanco was due to exposure to saxitoxin from the phytoplankton *Gymmodinium catenatum*, a product of harmful algal blooms (CMS 2005). A virus similar to canine distemper and morbillivirus, possibly transmitted from cetaceans, has also been found in monk seals from Cabo Blanco and implicated in deaths there (CMS 2005; Osterhaus et al. 1998; Van de Bildt et al. 2001; Van de Bildt et al. 1999). The virus has also been found in the Greek sub-population (Van de Bildt et al. 2000; Van de Bildt et al. 1999).

Additionally, genetic diversity in the Cabo Blanco sub-population is one of the lowest of any pinniped population studied, losing roughly 53% of its heterozygosity (CMS 2005; Pastor et al. 2007; Pastor et al. 2004). However, inbreeding does not yet appear to be a problem (CMS 2005; Pastor et al. 2007). Genetic diversity in the Greek population is also low (Pastor et al. 2007).

4.2.5.8 Anthropogenic threats.

Mediterranean monk seals were historically numerous but have been reduced to a small fraction of their former abundance and range due to human exploitation over the past 600 years (Brito et al. 2006; Sergeant et al. 1978). Occurrence on open beaches generally ended due to exploitation during the 15th and 16th centuries (CMS 2005). Since exploitation for human use, monk seals were killed incidentally or intentionally as part of fisheries activities, with numerous reports of individuals killed throughout the present range of the species (CMS 2005; Gonzalez et al. 1993; Panou et al. 1993), notably in association with gill and bottom trawl nets (Gonzalez and Larrinoa 2013; Guclusoy 2008a) but also hand lines in Madeira (Hale et al. 2011). Sub-adults may be the age class most affected by entanglement (Karamanlidis et al. 2008).

The primary mortality threat to Mediterranean monk seals in Greek waters has been targeted and incidental killing and habitat destruction (Androukaki et al. 1998; Danyer et al. 2013; Guclusoy et al. 2004; MOm 2014a; Panou et al. 1993). For adults and juveniles, deliberate killing was found to be the case of death of 43% of monk seals in cases examined from 1973-1996 in Greek waters, with an additional 13% caused incidental to fishing operations and only 23% stemming from natural causes (Androukaki et al. 1998; Cebrian et al. 1998). In contrast to this, natural reasons were the cause of death for 92% of pups. A rehabilitation center was established in 1990 specifically to aid in the survival of stranded or injured individuals which has treated and released several individuals (Androukaki et al. 2001).

Organochlorine pollutants have been identified in Mediterranean monk seal subpopulations (Borrell et al. 1993; Borrell et al. 1997). As with other marine mammals, levels increase with age, with females showing reductions at sexual maturity as pollutant burdens are shared with offspring (Borrell et al. 1993). Measured levels can meet or exceed levels which, in other seal species, can be carcinogenic, and have led to female reproductive impairment and reduced immune competence (Borrell et al. 1993; Borrell et al. 1997). Based upon different congener proportions, Atlantic sub-populations are exposed to more contaminants of industrial origin, whereas Mediterranean sub-populations are exposed to more agricultural origin pollution (Borrell et al. 2007). Halogenated pollutants have also been identified in Mediterranean monk seal blubber, although the implications of this are unknown (Hiebl et al. 2006). Arsenic and selenium levels in monk seals from Greek water are high enough to disrupt endocrine or reproductive function in gray seals (Zaccaroni et al. 2014).

4.2.6 Loggerhead Sea Turtle

4.2.6.1 Distribution

Loggerheads are circumglobal occurring throughout the temperate and tropical regions of the Atlantic, Pacific, and Indian oceans. Loggerheads are the most abundant species of sea turtle found in Mediterranean waters (Broderick et al. 2002b), and found throughout the Mediterranean, although nesting mostly occurs in the eastern Mediterranean basin (Camiñas and De Málaga 2004; Conant 2009).

4.2.6.2 Population Designations

Five groupings represent loggerhead sea turtles by major sea or ocean basin: Atlantic, Pacific, and Indian oceans, as well as Caribbean and Mediterranean seas. As with other sea turtles, populations are frequently divided by nesting aggregation (Hutchinson and Dutton 2007). On March 16, 2010, the NMFS proposed to designate 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).

The loggerhead Mediterranean Sea DPS is genetically distinct, and testing performed on mitochondrial DNA indicates that there are four independent subpopulations based on different nesting groups within the Mediterranean: 1) Mainland Greece and the adjoining Ionian Islands, 2) eastern Turkey, 3) Israel, and 4) Cyprus (Conant 2009).

4.2.6.3 Reproduction and Growth

Loggerhead nesting is confined to lower latitudes temperate and subtropic zones but absent from tropical areas (NMFS and USFWS 1991b; NRC 1990b; Witherington et al. 2006b). The life cycle of loggerhead sea turtles can be divided into seven stages: eggs and hatchlings, small juveniles, large juveniles, sub-adults, novice breeders, first year emigrants, and mature breeders (Crouse et al. 1987). Hatchling loggerheads migrate to the ocean (to which they are drawn by near ultraviolet light Kawamura et al. 2009), where they are generally believed to lead a pelagic existence for as long as 7-12 years (NMFS 2005a). Loggerheads in the Mediterranean, similar to those in the Atlantic, grow at roughly 11.8 cm/yr for the first six months and slow to roughly 3.6

cm/yr at age 2.5-3.5. As adults, individuals may experience a secondary growth pulse associated with shifting into neritic habitats, although growth is generally monotypic (declines with age Casale et al. 2009a; Casale et al. 2009b). Individually-based variables likely have a high impact on individual-to-individual growth rates (Casale et al. 2009b). At 15-38 years, loggerhead sea turtles become sexually mature, although the age at which they reach maturity varies widely among populations (Casale et al. 2009b; Frazer and Ehrhart 1985; NMFS 2001a; Witherington et al. 2006). For loggerheads in the Mediterranean, the minimum age at maturity in 15-16 years (Margaritoulis 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 1988; NMFS and USFWS 1998d). Females usually breed every 2-3 years, but can vary from 1-7 years (Dodd 1988; Richardson et al. 1978). Female loggerheads in Cyprus had a nesting interval of 2 years (Broderick et al. 2002b). Females lay an average of 1.8-2.2 nests per season, and the breeding season is from late May to mid-August (Broderick et al. 2002b). 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 Mediterranean is confined almost exclusively to the eastern basin, with major nesting sites occurring in Cyprus, Turkey and Greece (Margaritoulis and Rees 2003). There are lesser nesting sites located in Libya, Egypt, Lebanon, Syria, southern Italy, Israel, Tunisia (Conant 2009). The highest level of nesting in the Mediterranean occurs in Greece, with an average of 3,050 nests per year (Margaritoulis et al. 2003); a total of 2,280-2,787 loggerheads are thought to nest in Cyprus annually (Broderick et al. 2002b). In Greece, the majority of loggerhead nesting occurs on the island of Zakynthos, part of the Ionian Islands on the western coast; nesting also occurs western and southern coasts of Greece and the island of Crete (Conant 2009) (Margaritoulis et al. 2003).

4.2.6.4 Migration and Movement

Loggerhead hatchlings migrate offshore and become associated with *Sargassum* spp. habitats, drift lines, and other convergence zones (Carr 1986). Loggerhead hatchlings disperse throughout the Mediterranean, with the location of their natal nesting beach influencing where they disperse. Hatchlings from the Levantine Basin (i.e., the eastern Mediterranean) and south-central Mediterranean tended to stay in that region, and similarly, hatchlings from the Ionian Sea dispersed through the Ionian and Adriatic Seas (Casale and Mariani 2014). After 14-32 years of age, they shift to a benthic habitat, where immature individuals forage in the open ocean and coastal areas along continental shelves, bays, lagoons, and estuaries (Bowen et al. 2004; NMFS 2001a). Adult loggerheads make lengthy migrations from nesting beaches to foraging grounds (TEWG 1998). After breeding, loggerheads overwinter in coastal areas with low (>15°C) water temperatures, undergoing long resting dives and foraging throughout the winter (Broderick et al. 2007) (Hochscheid et al. 2007).

Loggerheads in the Mediterranean have been the subject of several telemetry studies. Adults and juveniles show a tendency to disperse widely, traveling from the Ionian Sea to the Adriatic and Aegean Seas, and some from Turkey to the Aegean Sea (Bentivegna et al. 2007; Casale et al. 2007b; Margaritoulis and Teneketzis 2003).

4.2.6.5 Gender, Age, and Survivorship

Although information on males is limited, several studies identified a female bias, although a single study has found a strong male bias to be possible (Dodd 1988; NMFS 2001a; Rees and Margaritoulis 2004). Additionally, little is known about longevity, although Dodd (1988) estimated the maximum female life span at 47-62 years. Heppell et al. (2003b) estimated annual survivorship to be 0.81 (southeast U.S. adult females), 0.78-0.91 (Australia adult females), 0.68-0.89 (southeast U.S. benthic juveniles, and 0.92 (Australia benthic juveniles). Survival rates for hatchlings during their first year are likely very low (Heppell et al. 2003a).

4.2.6.6 Feeding

Loggerhead sea turtles are omnivorous and opportunistic feeders through their lifetimes (Parker et al. 2005). Hatchling loggerheads feed on macroplankton associated with *Sargassum* spp. communities (NMFS and USFWS 1991b). Pelagic and benthic juveniles forage on crabs, mollusks, jellyfish, and vegetation at or near the surface (Dodd 1988; Wallace et al. 2009). 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 (Bolten 2003; 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). Significant foraging areas in the Mediterranean include areas in the southwestern Aegean Sea (e.g., Amvarkikos Gulf, Argolikos Bay, Saronikos Bay), around Crete, in the southeastern Aegean Sea, and in the northern Aegean Sea (Casale and Margaritoulis 2010).

4.2.6.7 Diving

Loggerhead diving behavior varies based upon habitat, with longer surface stays in deeper habitats than in coastal ones. Off Japan, dives were shallower than 30 m (Sakamoto et al. 1993). Routine dives can last 4–172 min (Byles 1988; Renaud and Carpenter 1994; Sakamoto et al. 1990). The maximum-recorded dive depth for a post-nesting female was over 230 m, although most dives are far shallower (9-21 m) (Sakamoto et al. 1990). Loggerheads tagged in the Pacific over the course of 5 months showed that about 70% of dives are very shallow (<5 m) and 40% of their time was spent within 1 m of the surface (Polovina et al. 2003; Spotila 2004b). During these dives, there were also several strong surface temperature fronts that individuals were associated with, one of 20° C at 28° N latitude and another of 17° C at 32° N latitude. Loggerheads in the eastern Mediterranean on overwintering sites will dive to depths of 4-24m, lasting >3 to 10.2 hours (Broderick et al. 2007; Hochscheid et al. 2007).

4.2.6.8 Vocalization and Hearing

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

4.2.6.9 Status and Trends

Loggerhead sea turtles were listed as threatened under the ESA of 1973 on July 28, 1978 (43 FR 32800). The NMFS recently determined that a petition to reclassify loggerhead turtles in the western North Atlantic Ocean as endangered may be warranted due to the substantial scientific and commercial information presented. Consequently, NMFS has initiated a review of the status of the species and is currently soliciting additional information on the species status and ecology, as well as areas that may qualify as critical habitat (73 FR 11849; March 5, 2008).

There is general agreement that the number of nesting females provides a useful index of the species’ population size and stability at this life stage, even though there are doubts about the ability to estimate the overall population size (Bjorndal et al. 2005). An important caveat for population trends analysis based on nesting beach data is that this may reflect trends in adult nesting females, but it may not reflect overall population growth rates well. Adult nesting females often account for less than 1% of total population numbers. The global abundance of nesting female loggerhead turtles is estimated at 43,320–44,560 (Spotila 2004a). Based on average number of documented nests in 2010, there are over 7,200 loggerhead nests a year in the Mediterranean, although some sites are showing a decline in the number of nests (Otero and Conigliaro 2012).

The potential for future decline for the Mediterranean Sea DPS is considered high, according to the NMFS Loggerhead Biological Review Team due to threats from fisheries interactions, pollution, vessel collisions, and habitat destruction of nesting beaches. Indices of future population health for the Mediterranean DPS were calculated at various maximum population growth rates, the index was zero (Conant 2009).

4.2.6.10 Natural Threats

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

4.2.6.11 Anthropogenic Threats

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

As is the case for loggerheads all over the world, the major factors inhibiting the recovery of the Mediterranean Sea DPS include mortalities caused by fishery interactions and degradation of nesting beaches. Juvenile and adult loggerheads are captured in trawls, pelagic and bottom longline, gill nets, trammel nets and driftnet fisheries in the Mediterranean (Casale et al. 2007a; Conant 2009; Laurent 2001; Laurent et al. 1996). Deliberate hunting of loggerheads for their meat, shells, and eggs has declined from previous exploitation levels, but still exists and hampers recovery efforts (Lino et al. 2010).

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. 2010a); many of these are expected to be loggerhead sea turtles.

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

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

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

Heavy metals, including arsenic, barium, cadmium, chromium, iron, lead, nickel, selenium, silver, copper, zinc, and manganese, have also been found in a variety of tissues in levels that increase with turtle size (Anan et al. 2001; Fujihara et al. 2003; Garcia-Fernandez et al. 2009; Gardner et al. 2006; Godley et al. 1999; Saeki et al. 2000; Storelli et al. 2008). These metals likely originate from plants and seem to have high transfer coefficients (Anan et al. 2001; Celik et al. 2006; Talavera-Saenz et al. 2007). Loggerhead sea turtles have higher mercury levels than any other sea turtle studied, but concentrations are an order of magnitude less than many toothed whales (Godley et al. 1999; Pugh and Becker 2001).

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

4.2.6.12 Critical Habitat

Critical habitat for loggerheads has been designated in the United States, including areas in North Carolina, South Carolina, Georgia, Florida, Louisiana and Texas.

4.2.7 Green Sea Turtle

4.2.7.1 Distribution

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

4.2.7.2 Population Designation

Populations are distinguished generally by ocean basin and more specifically by nesting location (Table 11). There are several green turtle nesting sites in the Mediterranean, primarily in Turkey and Cyprus, with smaller nesting sites in Israel, Syria, Egypt and Lebanon (Kasparek et al. 2001) (Caminas 2004) (Seminoff 2015). The sub-population of green turtles in the Mediterranean is genetically distinct, with additional genetic variation amongst the various nesting beaches (Bagda et al. 2012; Encalada et al. 1996). Turkey hosts the largest number of nestings for green turtles in the Mediterranean (61%), with nesting sites in Cyprus also comprising a substantial number as well (Canbolat 2004). Limited green turtle nesting occurs in Greece, with one nest per year reported in 2007 (Casale and Margaritoulis 2010).

Table 11. 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. 2002a)

Cyprus	121-127 AF	(Broderick et al. 2002a)
Israel / Palestine	1-3 AF	(Kuller 1999)
Syria	100 AN	(Rees et al. 2005)
Western Indian Ocean		
Eparces Islands	2,000-11,000 AF	(Le Gall et al. 1986)
Comoros Islands	5,000 AF	S. Ahamada, pers. comm. 2001
Seychelles Islands	3,535-4,755 AF	J. Mortimer, pers. comm. 2002
Kenya	200-300 AF	(Okemwa and Wamukota 2006)
Northern Indian Ocean		
Ras al Hadd, Oman	44,000 AN	S. Al-Saady, pers. comm. 2007
Sharma, Yemen	15 AF	(Saad 1999)
Karan Island, Saudi Arabia	408-559 AF	(Pilcher 2000)
Jana and Juraid Islands, Saudi Arabia	643 AN	(Pilcher 2000)
Hawkes Bay and Sandspit, Pakistan	600 AN	(Asrar 1999)
Gujarat, India	461 AN	(Sunderraj et al. 2006)
Sri Lanka	184 AF	(Kapurisinghe 2006)
Eastern Indian Ocean		
Thamihla Kyun, Myanmar	<250,000 EH	(Thorbjarnarson et al. 2000)
Pangumbahan, Indonesia	400,000 EH	(Schulz 1987)
Suka Made, Indonesia	395 AN	C. Limpus, pers. comm. 2002
Western Australia	3,000-30,000 AN	R. Prince, pers. comm. 2001
Southeast Asia		
Gulf of Thailand	250 AN	Charuchinda pers. comm. 2001
Vietnam	239 AF	(Hamann et al. 2006a)
Berau Islands, Indonesia	4,000-5,000 AF	(Schulz 1984)

Turtle Islands, Philippines	1.4 million EP	(Cruz 2002)
Sabah Turtle Islands, Malaysia	8,000 AN	(Chan 2006)
Sipadan, Malaysia	800 AN	(Chan 2006)
Sarawak, Malaysia	2,000 AN	(Liew 2002)
Enu Island (Aru Islands)	540 AF	Dethmers, in preparation
Terengganu, Malaysia	2,200 AN	(Chan 2006)
Western Pacific Ocean		
Heron Island and southern Great Barrier Reef areas, Australia	5,000-10,000 AF	(Maison et al. 2010)
Raine Island and northern Great Barrier Reef areas, Australia	10,000-25,000 AF	(Limpus et al. 2003) (Maison et al. 2010)
Coringa-Herald National Nature Reserve, Australia	1,445 AF	(Maison et al. 2010)
Guam	45 AF	(Cummings 2002)
Phoenix Islands, Kiribati	100-300 AF	(Maison et al. 2010)
Ogasawara Islands, Japan	500 AF	(Chaloupka et al. 2007)
Micronesia	500-1,000 AF	(Maison et al. 2010)
Marshall Islands	100-500 AF	(Maison et al. 2010)
New Caledonia	1,000-2,000 AF	(Maison et al. 2010)
Central and Eastern Pacific Ocean		
French Frigate Shoals, Hawaii	400 AF	(Balazs and Chaloupka 2006)
Michoacán, Mexico	1,395 AF	C. Delgado, pers. comm. 2006
Central American Coast	184-344 AN	(López and Arauz 2003)
Galapagos Islands, Ecuador	1,650 AF	(Zárate et al. 2006)

4.2.7.3 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. 2000a; Seminoff et al. 2002b). If individuals do not feed sufficiently, growth is stunted and apparently does not compensate even when greater-than-needed resources are available (Roark et al. 2009). In general, there is a tendency for green sea turtles to exhibit monotonic growth (declining growth rate with size) in the Atlantic and non-monotonic growth (growth spurt in mid-size classes) in the Pacific, although this is not always the case (Balazs and Chaloupka 2004; Chaloupka and Musick 1997; Seminoff et al. 2002b). It is estimated that green sea turtles reach a maximum size just under 100 cm in carapace length (Tanaka 2009). A female-bias has been identified from studies of green sea turtles (Wibbels 2003).

Consistent with slow growth, age-to-maturity for green sea turtles appears to be the longest of any sea turtle species and ranges from ~20-40 years or more (Chaloupka et al. 2004; Chaloupka and Musick 1997; 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. 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).

There is some information available regarding green turtle reproduction in the Mediterranean. In a study examining the nesting behavior of green turtles in northern Cyprus, the median interval between nesting seasons was three years, with a clutch frequency of 2.9-3.1 clutches per female (Broderick et al. 2002b). An estimated 350-1750 clutches are laid annually throughout the Mediterranean (Kasparek et al. 2001). Nesting occurs from late May to mid-August (Broderick et al. 2002b).

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. 2002a).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 sub-adults 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).

4.2.7.4 Migration and Movement

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

4.2.7.5 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 juveniles green sea turtles along the Pacific Baja coast (Senko et al. 2010b).

Green turtles are year-round residents in the eastern Mediterranean, staying mostly in the coastal, neritic environment of the eastern Levantine Basin, near Cyprus, Turkey, and Egypt (Broderick et al. 2002b). Young green turtles appear to overwinter in northern Africa, Libya, and Egypt, and to a lesser extent in southern Turkey (Broderick et al. 2002b) (Godley et al. 2002b). In a hatching dispersal simulation of green turtles from nesting sites in Turkey and Cyprus, hatchlings mostly stayed in the Levantine Basin, while hatchlings from the Ionian area dispersed into the Ionian, Adriatic and south-central Mediterranean (Casale and Mariani 2014). Foraging areas for juveniles seem to be based upon seagrass and macroalgae abundance, such as along the coast of Turkey, Egypt, Rhode Island and Peloponnesus in Greece (Godley et al. 2002b) (Broderick et al. 2007; Margaritoulis et al. 2011). However, green sea turtles may also occur in offshore regions, particularly during migration and development.

4.2.7.6 Feeding

While offshore and sometimes in coastal habitats, green sea turtles are not obligate plant-eaters as widely believed, and instead consume invertebrates such as jellyfish, sponges, sea pens, and pelagic prey (Godley et al. 1998; Hatase et al. 2006; Heithaus et al. 2002; Parker and Balazs in press; Seminoff et al. 2002a). A shift to a more herbivorous diet occurs when individuals move into neritic habitats, as vegetable 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).

4.2.7.7 Diving

Based on the behavior of post-hatchlings and juvenile green turtles raised in captivity, we presume that those in pelagic habitats live and feed at or near the ocean surface, and that their dives do not normally exceed several meters in depth (Hazel et al. 2009; NMFS and USFWS 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 sub-adults routinely dive to 20 m for 9-23 min, with a maximum recorded dive of over 1 h (Brill et al. 1995; I-Jiunn 2009). Green sea turtles along Taiwan may rest during long, shallow dives (I-Jiunn 2009). Dives by females may be shorter in the period leading up to nesting (I-Jiunn 2009).

4.2.7.8 Vocalization and Hearing

Although very limited information is available regarding green turtle hearing, it is one of the few sea turtle species that have been studied. Based upon auditory brainstem responses green sea turtles have been measured to hear in the 50-1600 Hz range (Dow et al. 2008) and 100-800 Hz (Bartol and Ketten 2006), although cochlear potential suggest a range between 60 and 1000 Hz (Ridgway et al. 1969a). Maximum sensitivity has been found to be 200-400 Hz for sub-adults and 600-700 for juveniles (Bartol and Ketten 2006; Ketten and Bartol 2006). This is supported by cochlear potential estimates of 300-500 Hz from Ridgway et al. (1969a). However, Dow et al. (2008) found best sensitivity between 50 and 400 Hz. Outside of this limited range, green turtles are much less sensitive to sound (Ridgway et al. 1969a). This is similar to estimates for loggerhead sea turtles, which have most sensitive hearing between 250-1,000 Hz, with rapid decline above 1,000 Hz (Moein Bartol et al. 1999).

4.2.7.9 Status and Trends

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

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

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

The status and trends for green turtles in the Mediterranean (and specifically Greece) are unknown (Casale and Margaritoulis 2010). Nesting data in Turkey and Cyprus (the location of the two most abundant nesting regions for the Mediterranean DPS) show no apparent downward trend (Seminoff 2015). The average number of green turtle nests is over 1,500 per year (based on 2010 data) (Otero and Conigliaro 2012).

4.2.7.10 Natural Threats

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

4.2.7.11 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 nesting females, and may evoke a change in the natural behaviors of adults and hatchlings (Ackerman 1997; Witherington et al. 2003; Witherington et al. 2007). The presence of lights on or adjacent to nesting beaches alters the behavior of nesting adults (Witherington 1992) and is often fatal to emerging hatchlings as they are attracted to light sources and drawn away from the water (Witherington and Bjorndal 1991). In addition to impacting the terrestrial zone, anthropogenic disturbances also threaten coastal marine habitats, particularly areas rich in seagrass and marine algae. These impacts include contamination from herbicides, pesticides, oil spills, and other chemicals, as well as structural degradation from excessive boat anchoring and dredging (Francour et al. 1999; Lee Long et al. 2000; Waycott et al. 2005). Ingestion of plastic and other marine debris is another source of morbidity and mortality (Stamper et al. 2009). Green sea turtles stranded in Brazil were all found to have ingested plastics or fishing debris (n=34), although mortality appears to have results 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).

Sea level rise may have significant impacts upon green turtle nesting on Pacific atolls. These low-lying, isolated locations could be inundated by rising water levels associated with global warming, eliminating nesting habitat (Baker et al. 2006; Fuentes et al. 2010a). Fuentes et al. (2010) predicted that rising temperatures would be a much greater threat in the long term to the hatching success of sea turtle turtles in general and green sea turtles along northeastern Australia particularly. Green sea turtles emerging from nests at cooler temperatures likely absorb more yolk that is converted to body tissue than do hatchlings from warmer nests (Ischer et al. 2009). Predicted temperature rises may approach or exceed the upper thermal tolerance limit of sea turtle incubation, causing widespread failure of nests (Fuentes et al. 2010a). 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).

4.2.7.12 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.

4.2.8 Leatherback Sea Turtle

4.2.8.1 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. Leatherback sea turtles are infrequent in the Mediterranean Sea, and those individuals are likely from the Atlantic Ocean nesting aggregation; regular nesting in the Mediterranean by leatherbacks has not been observed (Caminas 2004).

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).

4.2.8.2 Distribution

Leatherbacks range farther than any other sea turtle species, having evolved physiological and anatomical adaptations that allow them to exploit cold waters (Frair et al. 1972; Greer et al. 1973; USFWS 1995). High-latitude leatherback range includes in the Atlantic includes the North and Barents Seas, Newfoundland and Labrador, Argentina, and South Africa (Goff and Lien 1988; Hughes et al. 1998; Luschi et al. 2003; Luschi et al. 2006; Márquez 1990; Threlfall 1978). Pacific ranges extend to Alaska, Chile, and New Zealand (Brito 1998; Gill 1997; Hodge and Wing 2000). Associations exist with continental shelf and pelagic environments and sightings occur in offshore waters of 7-27˚ C (CETAP 1982). Juvenile leatherbacks usually stay in warmer, tropical waters >21˚ C (Eckert 2002). Males and females show some degree of natal homing to annual breeding sites (James et al. 2005).

A review of reports of leatherback sea turtles in the Mediterranean (mostly from 1981-2000) found 411 for all of the Mediterranean, with most records coming from waters west of Italy (Casale et al. 2003). From 1982-1984, 11 leatherbacks were reported in the Aegean Sea (Margaritoulis 1986), and five were reported in Greece from 1992-2000 (Panagopoulos et al. 2003a).

4.2.8.3 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.

4.2.8.4 Habitat

Leatherbacks occur throughout marine waters, from nearshore habitats to oceanic environments (Grant and Ferrell 1993; Schroeder and Thompson 1987; Shoop and Kenney 1992; Starbird et al. 1993). Movements are largely dependent upon reproductive and feeding cycles and the oceanographic features that concentrate prey, such as frontal systems, 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.

4.2.8.5 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 have been observed in the Mediterranean year-round, with more frequent sightings in summer months (Casale et al. 2003).

4.2.8.6 Sex Ratio

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

4.2.8.7 Feeding

Leatherbacks may forage in high-invertebrate prey density areas formed by favorable oceanographic features (Eckert 2006; Ferraroli et al. 2004). Although leatherbacks forage in coastal waters, they appear to remain primarily pelagic through all life stages (Heppell et al. 2003b). 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).

4.2.8.8 Vocalization and Hearing

Sea turtles are low-frequency hearing specialists, typically hearing frequencies from 30 to 2,000 Hz, with a range of maximum sensitivity between 100 and 800 Hz (Bartol et al. 1999; Lenhardt 1994b; Lenhardt 2002; Moein Bartol and Ketten 2006; Ridgway et al. 1969b). 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 1994b).

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).

4.2.8.9 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 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 2004a). The species as a whole is declining and local populations are in danger of extinction (NMFS 2001a; NMFS 2001b) (Table 12).

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).

Table 12. 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.

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	▼	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)

[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).

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).

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.

4.2.8.10 Natural Threats

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

4.2.8.11 Anthropogenic Threats

Leatherback nesting and marine environments are facing increasing impacts through widespread development and tourism along nesting beaches (Hamann et al. 2006b; 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. 1997c). 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). Plastic ingestion is very common in leatherbacks and can block gastrointestinal tracts leading to death (Mrosovsky et al. 2009). Along the coast of Peru, 13% of 140 leatherback carcasses were found to contain plastic bags and film (Fritts 1982). 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. 2010b); many of these turtles are expected to be leatherbacks. Currently, the U.S. tuna and swordfish longline fisheries managed under the HMS FMP are estimated to capture 1,764 leatherbacks (no more than 252 mortalities) for each 3-year period starting in 2007 (NMFS 2004). While 2010 total estimates are not yet available, in 2009, 285.8 (95% CI: 209.6-389.7) leatherback sea turtles are estimated to have been taken in the longline fisheries managed under the HMS FMP based on the observed takes (Garrison and Stokes 2010). Observer coverage for this period ranged from 54 to 92%. Trinidad and Tobago's Institute for Marine Affairs estimated that more than 3,000 leatherbacks were captured incidental to gillnet fishing in the coastal waters of Trinidad in 2000. Half or more of the gravid turtles in Trinidad and Tobago waters may be killed (Lee Lum 2003), though many of the turtles do not die as a result of drowning, but rather because the fishermen butcher them in order to get them out of their nets (NMFS 2001a).

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).

4.2.8.12 Critical Habitat

Critical habitat has been designated to leatherback sea turtles in the Atlantic in the U.S. Virgin Islands (44 FR 17710) and in the Pacific along the U.S. West coast (77 FR 4170).

5 ENVIRONMENTAL BASELINE

The “environmental baseline” includes the past and present impacts of all Federal, state, or private actions and other human activities in the action area, the anticipated impacts of all proposed Federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of state or private actions which are contemporaneous with the consultation in process (50 CFR 402.02).

5.1 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). 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). North Atlantic and Pacific sea surface temperatures have shown trends in being anonymously warm in recent years (Blunden and Arndt 2013). The ocean along the U.S. eastern seaboard is also much saltier than historical averages (Blunden and Arndt 2013).

The direct effects of climate change will result in increases in atmospheric temperatures, changes in sea surface temperatures, patterns of precipitation, and sea level. As described in the *Status of Listed Resources* for each sea turtle species, temperature regimes are generally leading towards female-biased nests. This can result in heavily feminized populations incapable of fertilization of available females (Laloë et al. 2014). This is not considered to be an imminent threat and presently has the advantage of shifting the natural rates of population growth higher (Laloë et al. 2014). 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 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 IPCC (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 under ice 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).

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 (Blunden and Arndt 2013; 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).

Changes in global climatic patterns will likely have profound effects on the coastlines of every continent by increasing sea levels and the intensity, if not the frequency, of hurricanes and tropical storms (Wilkinson and Souter 2008). A half degree Celsius increase in temperatures during hurricane season from 1965-2005 correlated with a 40% increase in cyclone activity in the Atlantic. Sea levels have risen an average of 1.7 mm/year over the 20th century due to glacial melting and thermal expansion of ocean water; this rate will likely increase. The current pace is nearly double this, with a 20-year trend of 3.2 mm/year (Blunden and Arndt 2013). This is largely due to thermal expansion of water, with minor contributions from melt water (Blunden and Arndt 2013). 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). In addition, flatter beaches preferred by smaller sea turtle species would be inundated sooner than would steeper beaches preferred by larger species (Hawkes et al. 2014). The loss of nesting beaches, by itself, would have catastrophic effects on sea turtle populations globally if they are unable to colonize new beaches that form or if the beaches do not provide the habitat attributes (sand depth, temperature regimes, refuge) necessary for egg survival. In some areas, increases in sea level alone may be sufficient to inundate sea turtle nests and reduce hatching success (Caut et al. 2009b). Storms may also cause direct harm to sea turtles, causing “mass” strandings and mortality (Poloczanska et al. 2009). Increasing temperatures in sea turtle nests alters sex ratios, reduces incubation times (producing smaller hatchling), and reduces nesting success due to exceeded thermal tolerances (Fuentes et al. 2009b; Fuentes et al. 2010b; 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 NAO, 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. 2002). This interaction affects sea surface temperatures, wind patterns, and oceanic circulation in the North Atlantic (Stenseth et al. 2002). The NAO shifts between positive and negative phases, with a positive phase having persisted since 1970 (Hurrell 1995). North Atlantic conditions experienced during positive NAO 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 NAO 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 NAO years (Conversi et al. 2001; Greene and Pershing 2004; Greene et al. 2003). 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. 2003).

5.2 Habitat Degradation

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

Natural sources of ambient noise include: wind, waves, surf noise, precipitation, thunder, and biological noise from marine mammals, fishes, and crustaceans. Anthropogenic sources of ambient noise include: transportation and shipping traffic, dredging, construction activities, geophysical surveys, and sonars. In general, it has been asserted that ocean background noise levels have doubled every decade for the last six decades in some areas, primarily due to shipping traffic (IWC 2004). The acoustic noise that commercial traffic contributes to the marine environment is a concern for listed species because it may impair communication between individuals (Hatch et al. 2008), 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 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). Microplastics have been detected in whale feeding grounds in the Mediterranean, placing baleen whales at risk of ingesting microplastics; the impacts of ingestion remain unknown (Deudero and Alomar 2015; Fossi 2015).

Ingestion of marine debris can have fatal consequences even for large whales as well as sea turtles. 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). Further incidents may occur but remain undocumented when carcasses do not strand.

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. One study found 37% of dead leatherback turtles had ingested various types of plastic (Mrosovsky et al. 2009). A Brazilian study found that 60% of stranded green sea turtles had ingested marine debris (primarily plastic and oil; (Bugoni et al. 2001)). Loggerhead sea turtles had a lesser frequency of marine debris ingestion. Plastic is possibly ingested out of curiosity or due to confusion with prey items; for example, plastic bags can resemble jellyfish (Milton and Lutz 2003). Marine debris consumption has been shown to depress growth rates in post-hatchling loggerhead sea turtles, elongating the time required to reach sexual maturity and increasing predation risk (McCauley and Bjorndal 1999). Sea turtles can also become entangled and die in marine debris, such as discarded nets and monofilament line (Laist et al. 1999; Lutcavage et al. 1997a; NRC 1990a; O'Hara et al. 1988). This fundamentally reduces the reproductive potential of affected populations, many of which are already declining.

5.3 Dredging

Marine dredging vessels are common within Mediterranean ports (Net 2011). 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. However, the construction and maintenance of federal navigation channels and dredging in sand mining sites have been identified as sources of sea turtle mortality and are currently being undertaken along the U.S. east coast, such as in Port Everglades, Florida. 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).

5.4 Seismic Surveys

Seismic surveys occur in the Mediterranean, and in Greek waters, for the purposes including oil and gas exploration and geological studies (Anonymous 1993; LGL 2015). A 3D seismic survey was conducted recently by Energean Oil and Gas in July 2015 for 23 days in the Gulf of Kavala, northeastern Greece (Gas 2015). The survey used eight streamers and covered an area of 340km^2. In addition, beginning in mid-November 2012 and extending through the end of January 2013, the Hellenic Republic Ministry of Environment, Energy and Climate Change conducted a seismic survey in the eastern Mediterranean Sea, mostly concentrated off the west coast of Greece[3].

5.5 Vessel Traffic

Vessel noise could affect marine animals in the study area. Shipping noise generally dominates ambient noise at frequencies from 20 to 300 Hz (Andrew et al. 2002; Hildebrand 2009; Richardson et al. 1995c). Background noise has increased significantly in the past 50 years as a result of increasing vessel traffic, and particularly shipping, with increases of as much as 12 dB in low frequency ranges; background noise may be 20 dB higher now versus preindustrial periods (Hildebrand 2009; Jasny et al. 2005; McDonald et al. 2006; NRC 1994; NRC 2003; NRC 2005; Richardson et al. 1995a). Over the past 50 years, the number of commercial vessels has tripled, carrying an estimated six times as much cargo (requiring larger, more powerful vessels) (Hildebrand 2009). Seismic signals also contribute significantly to the low frequency ambient sound field (Hildebrand 2009). Baleen whales may be more sensitive to sound at those low frequencies than are toothed whales. 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 (Boness et al. 2013; Holt et al. 2013; Parks 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).

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[3] http://www.pgs.com/Pressroom/Calendar_of_Events/Campaigns/2012/MultiClient-Newsletter/Middle-East-Newsletter/PGS-MultiClient-enters-Greece-with-GeoStreamer-GS/

5.6 Military Training Exercises

Military training occurs in the Mediterranean Sea, with a recent naval exercise conducted by Russia and China in the Black Sea in May 2015[4].

5.7 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.

Fishery interaction remains a major factor in sea turtle recovery and, frequently, the lack thereof. Wallace et al. (2010b) 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 turtle excluder devices and other bycatch reduction devices have significantly reduced the level of bycatch to sea turtles and other marine species in U.S. waters, mortality still occurs.

5.8 Invasive Species

Invasive species have been referred to as one of the top four threats to the world’s oceans consistently ranked behind habitat degradation and alteration (Pughiuc 2010; Raaymakers 2003; Raaymakers and Hilliard 2002; Terdalkar et al. 2005; Wambiji et al. 2007). In most cases, habitat is directly affected by human alterations, such as hydromodification, mining, dredging, drilling, and construction. However, invasive species, facilitated by human commerce, have the ability to directly alter ecosystems upon which listed species rely.

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[4] http://www.ibtimes.com/russia-china-begin-mediterranean-military-exercises-black-sea-port-visit-1916868

Invasive species are a major threat to many ESA-listed species. For species listed by the United States Fish and Wildlife Service (USFWS), 26% were listed partially because of the impacts of invasive species and 7% were listed because invasive species were the major cause of listing (Anttila et al. 1998). Pimentel et al. (2004) found that roughly 40% of listed species are at risk of becoming endangered or extinct completely or in part due to invasive species, while Wilcove et al. (1998) found this to be 49%, with 27% of invertebrates, 37% of reptiles, 53% of fishes, and 57% of plants imperiled partly or wholly due to non-native invasions. In some regions of the world, up to 80% of species facing extinction are threatened by invasive species (Pimentel et al. 2004; Yan et al. 2002). Clavero and Garcia-Bertro (2005) found that invasive species were a contributing cause to over half of the extinct species in the International Union for the Conservation of Nature database; invasive species were the only cited cause in 20% of those cases. Richter et al. (1997) identified invasive species as one of three top threats to threatened and endangered freshwater species in the U.S. as a whole.

5.9 Diseases

The impacts of introduced pathogens in the aquatic environment has been poorly explored and we likely know very little about the true frequency and significance of pathogen invasions (Drake et al. 2001). Pathogens are known to have adverse effects to invertebrate communities. Molluscs such as black and white abalone seem to be particularly sensitive to pathogens. Various species of the genus *Vibrio*, known to cause cholera in humans, white pox and white plague type II diseases in corals, and mortality in abalone of the same genus as black and white abalone, have been identified in ports and ballast water of vessels (Aguirremacedo et al. 2008; Anguiano-Beltrán et al. 1998; Ben-Haim and Rosenberg 2002). Oyster species have sustained several outbreaks from invasive pathogens, including *Haplosporidium nelsoni* (the cause of MSX disease, which Chesapeake Bay eastern oysters have shown 75-92% mortality to) and *Perkinsus marinus* (the cause of Dermo disease) in California, eastern North America, and Europe (Andrews 1984; Burreson and Ford 2004; Burreson et al. 2000; Ford and Haskin 1982; Renault et al. 2000), Bonamia ostreae in Europe (Ciguarria and Elston 1997; Van Banning 1987), and in the northeastern U.S., respectively (Ford 1996). Although specific instances of sea turtle pathogen transference via invasive species are not documented, their spread into new areas are easily possible, particularly given environmental perturbations and naïve individuals in receiving habitats.

5.10 Ship Strikes

Ship-strike is a significant concern for the recovery of 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. 2007a). 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. 2011).

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.

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 2004a). 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. 2007b). 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 2001; NMFS 2005b). 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 2001). 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).[5] 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. 2011).

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[5] 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. 2001. 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..

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). A more recent examination of 28 sperm whale strandings in Greece between 1992 and 2014 showed that 12 showed clear evidence of vessel collision, prompting the authors to recommend consideration of changes in shipping routes to reduce risk of sperm whale ship strike in the Hellenic Trench (Frantzis et al. 2015). Additional efforts have been made to identify critical areas for fin whales in the Mediterranean to mitigate effects of ship strikes (Panigada et al. 2015).Sound propagation modeling showed that the level of sound received by whales depends on season and dive cycles, and the speed of the vessel approaching the whale reducing its ability to out-maneuver the vessel (Gannier and Marty 2015).

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.

5.11 Commercial Whaling

Large whale population numbers in the action areas have historically been impacted by commercial exploitation, mainly in the form of whaling. Between 1969-1990, 14 fin whales were captured in coastal fisheries off Newfoundland and Labrador; of these, seven are known to have died because of capture (Lien 1994; Perkins and Beamish 1979).

5.12 Scientific and Research Activities

Scientific research permits issued by the NMFS currently authorize studies of 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.

These research activities were not expected to jeopardize the survival or recovery of ESA-listed species and were largely anticipated to have short-term behavioral or stress effects to impacted individuals.

Authorized research on ESA-listed sea turtles includes capture, handling, and restraint, satellite, sonic, and passive integrated transponder tagging, blood and tissue collection, lavage, ultrasound, captive experiments, laparoscopy, and imaging. Research activities involve "takes" by harassment, with some resulting mortality. There have been numerous permits[6] issued since 2009 under the provisions of both the MMPA and ESA authorizing scientific research on marine mammals and sea turtles all over the world, including for research in the Mediterranean. The consultations which took place on the issuance of these ESA scientific research permits each found that the authorized activities would have no more than short-term effects and would not result in jeopardy to the species or adverse modification of designated critical habitat.

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.

5.13 Physical and Oceanographic Features

The presence of key habitat features, such as shelter or foraging opportunities, are the primary reasons why listed individuals occur where they do. In the marine environment, this is fundamentally built upon local physical and oceanographic features that influence the marine environment. As such, we describe the physical and oceanographic environment here to establish a rationale for why listed species occur in the action area at the levels we observe or expect. This does not represent a stressor, but is instead an underlying principle for establishing why effects are what we expect them to be.

5.14 Impacts of the Environmental Baseline on ESA-listed Species

Listed resources are exposed to a wide variety of past and present state, Federal or private actions and other human activities that have already occurred or continue to occur in the action

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[6]. See https://apps.nmfs.noaa.gov/index.cfm for additional details.

area. Any foreign projects in the action area that have already undergone formal or early section 7 consultation, and state or private actions that are contemporaneous with this consultation also impact listed resources. However, the impact of those activities on the status, trend, or the demographic processes of threatened and endangered species remains largely unknown. To the best of our ability, we summarize the effects we can determine based upon the information available to us in this section.

5.14.1 Marine Mammals

Climate change has wide-ranging impacts, some of which can be experienced by ESA-listed whales in the action area. Climate change has been demonstrated to alter major current regimes and may alter those in the action area as they are studied further (Johnson et al. 2011; Poloczanska et al. 2009). Considering the sensitivity that North Atlantic right whales have to warm water temperatures during their southbound migration, warming water temperatures may delay their migratory movements. The availability and quality of prey outside the action area in northern feeding areas can also influence the body condition of individuals in the action area, and potentially reduce the number of individuals that undertake migration through the action area.

Effects from anthropogenic acoustic sources, whether they are vessel noise, seismic sound, military activities, oil and gas activities, construction, or wind energy, could also have biologically significant impacts to ESA-listed whales in the action area. These activities increase the level of background noise in the marine environment, making communication more difficult over a variety of ranges. We expect that this increased collective noise also reduces the sensory information that individuals can gather from their environment; an important consideration for species that gather information about their environment primarily through sound. At closer ranges to some of anthropogenic sound sources, behavioral responses also occur, including deflecting off migratory paths and changing vocalization, diving, and swimming patterns. At even higher received sound levels, physiological changes are likely to occur, including temporary or permanent loss of hearing and potential trauma of other tissues. Although this exposure is a small fraction of the total exposure individuals receive, it is believed expected to occur in rare instances.

High levels of morbidity and mortality occur as a result of ship strike (particularly for North Atlantic right whales and humpback whales) and entanglement in fishing gear (right whales). Ship-strike and entanglement occur broadly along the U.S. East Coast, including (in all likelihood) in the action area itself. These two factors are the greatest known source of mortality and impairment to recovery for North Atlantic right whales and represent known mortality sources for all other ESA-listed whales in the action area. Reductions in speed through portions of the action area as well as seasonal or brief closings of areas to fishing are underway to reduce these impacts, but data are not yet available to demonstrate the long-term effectiveness of these strategies. However, these measures are likely reducing the severity and frequency of these interactions.

Authorized research on ESA-listed whales can have significant consequences for these species, particularly when viewed in the collective body of work that has been authorized. Researchers have noted changes in respiration, diving, swimming speed, social exchanges, and other behavior correlated with the number, speed, direction, and proximity of vessels. Responses were different depending on the age, life stage, social status of the whales being observed (i.e., males, cows with calves) and context (feeding, migrating, etc.). Beale and Monaghan (2004) concluded that the significance of disturbance was a function of the distance of humans to the animals, the number of humans making the close approach, and the frequency of the approaches. These results would suggest that the cumulative effects of the various human activities in the action area would be greater than the effects of the individual activity. Several investigators reported behavioral responses to close approaches that suggest that individual whales might experience stress responses. Baker et al. (1983) described two responses of whales to vessels, including: (1) “horizontal avoidance” of vessels 2,000 to 4,000 meters away characterized by faster swimming and fewer long dives; and (2) “vertical avoidance” of vessels from 0 to 2,000 meters away during which whales swam more slowly, but spent more time submerged. Watkins et al. (1981) found that both fin and humpback whales appeared to react to vessel approach by increasing swim speed, exhibiting a startled reaction, and moving away from the vessel with strong fluke motions. Other researchers have noted changes in respiration, diving, swimming speed, social exchanges, and other behavior correlated with the number, speed, direction, and proximity of vessels. Results were different depending on the social status of the whales being observed (single males when compared with cows and calves), but humpback whales generally tried to avoid vessels when the vessels were 0.5 to 1.0 kilometer from the whale. Smaller pods of whales and pods with calves seemed more responsive to approaching vessels (Bauer 1986; Bauer and Herman 1986). These stimuli are probably stressful to the humpback whales in the Action Area, but the consequences of this stress on the individual whales remains unknown (Baker and Herman 1987; Baker et al. 1983). Studies of other baleen whales, specifically bowhead and gray whales, document similar patterns of behavioral disturbance in response to a variety of actual and simulated vessel activity and noise (Malme et al. 1983; Richardson et al. 1985). For example, studies of bowhead whales revealed that these whales oriented themselves in relation to a vessel when the engine was on, and exhibited significant avoidance responses when the vessel’s engine was turned on even at a distance of about 900 m (3,000 ft). Jahoda et al. (2003) studied the response of 25 fin whales in feeding areas in the Ligurian Sea to close approaches by inflatable vessels and to biopsy samples. They concluded that close vessel approaches caused these whales to stop feeding and swim away from the approaching vessel. The whales also tended to reduce the time they spent at surface and increase their blow rates, suggesting an increase in metabolic rates that might indicate a stress response to the approach. In their study, whales that had been disturbed while feeding remained disturbed for hours after the exposure ended. They recommended keeping vessels more than 200 meters from whales and having approaching vessels move at low speeds to reduce visible reactions in these whales. Although these responses are generally ephemeral and behavioral in nature, populations within the action area can be exposed to several thousand instances of these activities per year, with some species having so many authorized activities that if they were all conducted, every individual in the population would experience multiple events. This can collectively alter the habitat use of individuals, or make what would normally be rare, unexpected effects (such as severe behavioral responses or infection from satellite or biopsy work) occur on a regular basis.

5.14.2 Sea Turtles

Several of the activities described in this *Environmental Baseline* have significant and adverse consequences for nesting sea turtle aggregations whose individuals occur in the Action Area. In particular, the commercial fisheries annually capture substantial numbers of green, Kemp's ridley, leatherback, and Northwest Atlantic loggerhead sea turtles.

Climate change has and will continue to impact sea turtles throughout the action area as well as throughout the range of the populations. Sex ratios of several species are showing a bias, sometimes very strongly, towards females due to higher incubation temperatures in nests. We expect this trend will continue and possibly may be exacerbated to the point that nests may become entirely feminized, resulting in severe demographic issues for affected populations in the future. Hurricanes may become more intense and/or frequent, impacting the nesting beaches of sea turtles and resulting in increased loss of nests over wide areas. Disease and prey distributions may well shift in response to changing ocean temperatures or current patterns, altering the morbidity and mortality regime faced by sea turtles and the availability of prey.

Although only small percentages of these sea turtles are estimated to have died as a result of their capture during research or incidental to fisheries, the actual number could be substantial if considered over the past 5 to 10 years. When we add the percentage of sea turtles that have suffered injuries or handling stress sufficient to have caused them to delay the age at which they reach maturity or the frequency at which they return to nesting beaches, the consequences of these fisheries on nesting aggregations of sea turtles would be greater than we have estimated.

Even with turtle excluder device measures in place, in 2002, NMFS (2002) expected these fisheries to capture about 323,600 sea turtles each year and kill about 5,600 (~1.7%) of the turtles captured. Loggerhead sea turtles account for most of this total: 163,000 captured, killing almost 4,000 (~2.5%) of them. These are followed by green sea turtles: about 18,700 captured with more than 500 (~2.7%) dying as a result of capture. Leatherback sea turtle interactions were estimated at 3,090 captures with 80 (~2.6%) deaths as a result (NMFS 2002b). Since 2002, however, effort in the Atlantic shrimp fisheries has declined from a high of 25,320 trips in 2002 to approximately 13,464 trips in 2009, roughly 47% less effort. Since sea turtle takes are directly linked to fishery effort, these takes are expected to decrease proportionately. However, hundreds to a possible few thousand sea turtle interactions are expected annually, with hundreds of deaths (NMFS 2012).

Recent data regarding the three largest subpopulations that comprise the Northwest Atlantic loggerhead DPS indicated either that these subpopulations do not show a nesting decline significantly different from zero (Peninsular Florida and The Greater Caribbean subpopulation) or are showing possible signs of stability in nest numbers (Northern subpopulation). These trends were recently declining. Additional mortalities each year along with other impacts remain a threat to the survival and recovery of this species and could slow recovery green, Kemp's ridley, leatherback, and Northwest Atlantic loggerhead sea turtles.

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

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

The proposed action involves non-lethal harassment of ESA listed cetaceans, pinnipeds, and sea turtles. The ESA does not define harassment nor has NMFS defined this term, pursuant to the ESA, through regulation. However, the MMPA defines harassment as "any act of pursuit, torment, or annoyance which has the potential to injure a marine mammal or marine mammal stock in the wild or has the potential to disturb a marine mammal or marine mammal stock in the wild by causing disruption of behavioral patterns, including, but not limited to, migration, breathing, nursing, breeding, feeding, or sheltering" [16 U.S.C. 1362(18)(A)]. The latter portion of this definition ("...causing disruption of behavioral patterns including...migration, breathing, nursing, breeding, feeding, or sheltering") is almost identical to the USFWS' regulatory definition of harass. For the following sections, we define harassment as "an intentional or unintentional act or omission that creates the probability of injury to an individual animal by disrupting one or more behavioral patterns essential to the animal's life history or its contribution to the population the animal represents." If we find that the proposed research causes behavioral disruptions that may result in animals that fail to feed or breed successfully, or die, then we will discuss the likely consequences of these disruptions for the population.

6.1 Stressors Associated with the Proposed Action

The potential stressors we expect to result from the proposed action are:

1. pollution by oil or fuel leakage;
2. acoustic interference from engine noise;
3. ship-strikes;
4. sound fields produced by airguns; and
5. sound fields produced by sub-bottom profiler or multibeam echosounder

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.

6.1.1 Pollution by Oil or Fuel Leakage

The potential for fuel or oil leakages is unlikely. Leaks 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 listed species directly or pose hazards to their food sources. Because the potential for fuel or oil leakage is extremely unlikely to occur, we find that the risk from this potential stressor is discountable. Therefore, we conclude that pollution by oil or fuel leakage is not likely to adversely affect ESA-listed marine mammals or sea turtles.

6.1.2 Disturbance from Engine Noise

The propulsion system of the *Langseth* and the chase vessel are 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 of noise generated by the *Langseth* 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 chase vessel would also not generate sufficient noise to significantly disturb ESA-listed marine mammals or sea turtles. Because the potential acoustic interference from engine noise would be undetectable or so minor that it could not be meaningfully evaluated, we find that the risk from this potential stressor is insignificant. Therefore, we conclude that acoustic interference from engine noise is not likely to adversely affect ESA-listed marine mammals or sea turtles.

6.1.3 Ship Strike

The *Langseth* and the chase vessel will be traveling at generally slow speeds, reducing the amount of noise produced by the propulsion system and the probability of a ship-strike (Kite-Powell et al. 2007; Vanderlaan and Taggart 2007). Our expectation of ship strike is discountably small 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). The same can be said for the chase vessel to be utilized. All factors considered, we have concluded the potential for ship strike from the research vessel or the chase vessel is highly improbable. Because the potential for ship strike is extremely unlikely to occur, we find that the risk from this potential stressor is discountable.

Therefore, we conclude that ship strike is not likely to adversely affect ESA-listed marine mammals or sea turtles.

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

6.2 Mitigation to Minimize or Avoid Exposure

NSF's proposed action includes the use of exclusion zones, protected species observers and operational shutdown in the presence of ESA-listed as species. The NMFS Permits and Conservation Division's proposed IHA would contain additional mitigation measures to minimize or avoid exposure. Both are described in the description of the action (see 2.1.6 and 2.2), and were considered throughout our analysis.

6.3 Exposure and Response Analysis

Exposure Analyses identify the physical, chemical, and biotic stressors produced by a proposed action that co-occur in space and time with ESA-listed species within the action area. The stressors identified for this proposed action that warrant further analysis are sound fields produced by airguns, and sound field produced by sub-bottom profiler or multibeam echosounder.

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.

6.3.1 Exposure Analysis

Although there are several acoustic and non-acoustic stressor associate with the proposed action, the stressor of primary concern is the acoustic impacts of airguns.

As part of the application for the IHA pursuant to the MMPA, the NSF provided an estimate of the number of marine mammals that would be exposed to levels of sound in which they would be considered "taken" during the proposed survey. NSF did not provide any take estimates from sound sources other than the airguns, although other equipment producing sound will be used during airgun operations (e.g., the multibeam echosounder and the sub-bottom profiler). In their FR Notice, the Permits and Conservation Division stated that they did not expect the sound emanating from the other equipment to exceed that of the airgun array. The Permits and Conservation Division did not expect additional exposure from sound sources other than the airguns. Since the sub-bottom profiler and the multibeam echosounder have a lower or roughly equivalent source output as the airgun array (**Table 3** and section 2.1.5), we agree with this assessment and similarly focus our analysis on exposure from the airgun array.

During the development of the IHA, the Permits and Conservation Division conducted an independent exposure analysis that was informed by comments received during the public comment period that was required on the proposed IHA and draft environmental assessment prepared pursuant to NEPA. In this section we describe both, the NSF and the NMFS analytical methods to estimate the number of ESA-listed species that might be exposed to the sound field and considered "taken" as required under the ESA.

For our ESA consultation, we evaluated both methods for estimating the number of ESA-listed individuals that would be "taken" relative to the definition of harassment discussed above. In most cases, we concur with the Permits and Conservation Divisions analysis. The NSF and NMFS analyses for each ESA-listed species are described below.

NMFS applies certain acoustic thresholds to help determine at what point during exposure to seismic airguns (and other acoustic sources) marine mammals are considered "harassed," under the MMPA. These thresholds are used to develop exclusion radii around a source and the necessary power-down or shut-down criteria to limit marine mammals and sea turtles' exposure to harmful levels of sound (NMFS 1995). The 160 dB re 1 μPa_{rms} distance is the distance at which MMPA take, by Level B harassment, is expected to occur, and the threshold at which the Permits and Conservation Division is proposing to issue take authorizations for marine mammals. The ESA does not define harassment nor has the NMFS defined the term pursuant to the ESA through regulation. The MMPA of 1972, as amended, defines harassment as any act of pursuit, torment, or annoyance which has the potential to injure a marine mammal or marine mammal population in the wild or has the potential to disturb a marine mammal or marine mammal population in the wild by causing disruption of behavioral patterns, including, but not limited to, migration, breathing, nursing, breeding, feeding, or sheltering [16 U.S.C. 1362(18)(A)]. The latter portion of this definition (that is, "...causing disruption of behavioral patterns including...migration, breathing, nursing, breeding, feeding, or sheltering") is similar to language in the USFWS's regulatory definition of "harass"[7] 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.

Airguns contribute a massive amount of anthropogenic energy to the world's oceans (3.9×10^{13} joules cumulatively), second only to nuclear explosions (Moore and Angliss 2006). Although most energy is in the low-frequency range, airguns emit a substantial amount of energy up to 150

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[7] 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)

kHz (Goold and Coates 2006). Seismic airgun noise can propagate substantial distances at low frequencies (e.g., Nieukirk et al. 2004).

6.3.1.1 NMFS Exposure Estimates and Territorial Seas

As discussed previously in section 2.3.2, the proposed seismic activities will occur inside and outside of the territorial seas of Greece. Under the ESA, we must consider the effects of the action throughout the entire action area in making our jeopardy determination. In this case, we only have jurisdiction to authorize incidental take of ESA-listed species outside the 6 n mi boundary. In estimating exposure, our calculations considered the exposure of all ESA-listed species within the entire action area (i.e., both in territorial seas and the high seas). Later in the opinion, based on these exposure numbers, we will detail the amount of incidental take of ESA-listed species that we have jurisdiction to authorize outside Greek territorial seas.

6.3.2 NSF Exposure Estimates

The NSF applied acoustic thresholds to determine at what point during exposure to seismic airguns marine mammals are “harassed,” based on definitions provided in the MMPA (65 FR 16374). The NSF concluded that ESA-listed whales and Mediterranean monk seals would be exposed to the seismic activities. The NSF did not provide estimates of sea turtle exposure. These thresholds were also used to develop exclusion radii around the acoustic source to determine appropriate power-down and shut-down procedures. The acoustic thresholds are described in Table 4. The NSF provided the predicted distances to which sound levels ≥ 190, 180, and 160 dB re 1 μPa_{rms} would be received (Level A and Level B harassment under the MMPA for pinnipeds and whales) are described above in Table 4. (NSF provided the predicted distances to which sound levels ≥ 166 dB re 1 μPa_{rms} would be received for sea turtles upon request).

The exposure analysis from NSF estimates the number of ESA-listed marine mammals likely to be exposed to received levels greater than 160 dB re 1 $\mu Pa_{rms,}$ which is a level that an adverse response by ESA-listed marine mammals is reasonable to expect. NSF provided estimates of exposure based on the potential exposures to sound levels received at the 160 dB re 1 $\mu Pa_{rms.}$ The rationale was that if exposed to sound levels from the proposed seismic activities, animals would move away from the comparatively larger 160 dB radius before received levels reached the higher sound levels of the 180 or 190 dB radii (LGL 2015).

The NSF exposure estimates were calculated based on the 1,105 km of seismic survey tracklines outside the 6 nautical mile territorial seas of Greece. The ensonified area was calculated to be 7,686 km^2, including a 25% contingency for the Santorini study area. Expected species density was multiplied by the ensonified area to arrive at numbers of animals exposed. NSF predicted that the ensonified area including overlap is 6.8 times the area excluding overlap. The Hellenic Trench survey line is a straight line that will be surveyed once, so there would be no overlapping ensonified areas in that instance.

There are no density estimates available for the action area in the waters surrounding Santorini or the Hellenic Trench subduction zone, so NSF chose whatever source was most appropriate to develop estimates of the number of individuals that could be exposed. For sperm and fin whales, NSF used density estimates developed by Laran et al. (2010) for the Ligurian Sea (approximately 1,000 miles northwest of Santorini). Because there are no density estimates for the Mediterranean for humpback whales or Mediterranean monk seals, NSF relied upon broader regional estimates of abundance. The regional estimate of abundance for Western North Atlantic humpback whales was based on Stevick et al. (2003b), and a status report published by the Hellenic Society for the Study and Protection of the Monk Seal (MOm 2009). NSF requested take authorization based on 1% of the regional population for each of these species. NSF did not request take authorization for sei whales, describing their occurrence in the Mediterranean as "extremely unlikely" (LGL 2015); see Table 14 for a summary of NSF's take request.

6.3.3 NMFS Exposure Estimates

The NMFS Permits and Conservation Division developed a daily ensonified area to apply to the species density estimates. Assuming that the *Langseth* was traveling at its fastest speed (4.5 kts) and a sea state at Beaufort 3 or less, the maximum amount of line kilometers that could be traveled in 24 hours is 200 km. They selected the first grouping of consecutive tracklines that had a total length of 200 km to represent the daily area that could be ensonified during seismic activities. An exclusion zone representing the predicted RMS distances (3.8 km) was then applied to this area; this buffer distance was provided by L-DEO. The Permits and Conservation Division estimated the daily ensonified area to be 3,809 km^2; this estimate does not account for overlap of ensonified areas. To account for additional contingency effort (25%), the Permits and Conservation Division included a 25% increase in the number of days (increasing from 16 to 20).

Exposure for each species for a single day was calculated by multiplying available species density by the daily ensonified area, and then the sum of those exposures over 20 days resulted in the final Permits and Conservation Division estimated take numbers. This calculation assumes 100% turnover of individuals within the ensonified area on a daily basis—that is, each individual exposed to the seismic activities is a unique individual. For species where the instance of exposure was less than one (humpback, and sei whales), the number of individuals exposed was increased to the mean group size based upon AMAPPS survey data. The results of the Permits and Conservation Division's calculations and their proposed take numbers is in Table 14, along with NSF's take authorization requests.

As described above, there is very little information available on the population abundance or density of the ESA-listed species within the action area. In most instances, we agreed with and adopted the Permits and Conservation Division's methodology for estimating exposure of ESA-listed marine mammals to the proposed action. The following section discusses each species and details the sources used to arrive at the number of exposures in our analysis, and how those estimates comport with the ones generated by the Permits and Conservation Division.

6.3.3.1 Exposure Estimates: Sperm and Fin Whales

There is no reliable population estimate for sperm whales in the Mediterranean, but it is thought to be a single population in the low hundreds; the local population size of sperm whales in the Hellenic Trench is estimated to be from 200-250 individuals (Frantzis et al. 2014). The population estimate for fin whales in the Mediterranean in about 5,000 adults (IUCN 2012), and there have been 36 sightings of fin whales in Greek waters (Notarbartolo-Di-Sciara et al. 2003).

Laran et al. (2010) estimated sperm and fin whale population density in the Ligurian Sea. Although the Ligurian Sea is some distance away from the action area, these estimates represent the best available density estimates for fin and sperm whales in the Mediterranean. The Permits and Conservation Division calculated exposure for fin and sperm whales by multiplying the winter density estimates developed by Laran et al. (Laran et al. 2010) by the daily ensonified area, and then the sum of those exposures over 20 days.

We concur with Permits and Conservation Division’s exposure estimates for sperm and fin whales (Table 14).

Fin whales of all age classes are likely to be exposed. Fin whales are expected to be feeding, traveling, or migrating in the area and some females of fin whales would have young-of-the-year accompanying them. We would normally assume that sex distribution is even for fin 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. For sperm whales, exposure for adult male sperm whales is expected to be lower than other age and sex class combinations.

6.3.3.2 Exposure Estimates: Sei and Humpback Whales

For sei whales, the Permits and Conservation Division adjusted the number of instances of exposure (1) to the mean group size, based on AMAPPS survey data. Given the relative rarity of sei whales in the Mediterranean, we believe this is an appropriate estimate of exposure for sei whales.

There is no density information available for humpback whales in the Mediterranean. For humpback whales, the Permits and Conservation Division adjusted the number of instances of exposure (3) to the mean group size, based on AMAPPS survey data, and multiplied the mean group size by the number of survey days. The Permits and Conservation Division is proposing to authorize 60 humpback whale takes.

In recent years, humpback whales have gone from extremely rare in the Mediterranean Sea (with two sightings in more than 100 years), to occasional, with nine new reports since 1990 (Frantzis et al. 2004b) (Frantzis et al. 2004). Two humpback whales were sighted in Greece in 2001 and 2002; the 2001 sighting was the first record of a humpback whale in the eastern Mediterranean basin. A third humpback was caught dead in fishing nets in 2004 (Frantzis et al. 2004b).

Humpback sightings are increasing in the Mediterranean, with 14 reported since 2001, with sightings more frequently recorded in the western Mediterranean (Panigada et al. 2014).

We recognize the possibility of humpback whales being exposed to the proposed seismic activities given the recent reports of humpbacks in the Mediterranean. However, based on the best information available, we do not believe it is likely that 60 humpback whales could be exposed to the proposed action. We believe it is appropriate to adjust the number of exposures of humpback whales to the mean group size of three based on AMAPPS survey data and the rarity of humpback whale sightings in the action area. We disagree with the Permits and Conservation Division's assertion that 60 humpback whales are likely to be exposed to the proposed seismic activities. Instead, we believe that, based on AMAPPS survey data and records of humpback whales in the Mediterranean, it is more likely that three humpback whales would be exposed to the proposed action.

Sei and humpback whales of all age classes are likely to be exposed. Whales are expected to be feeding, traveling, or migrating in the area and some females of all whale species would have young-of-the-year accompanying them. We would normally assume that sex distribution is even for sei and humpback whales and sexes are exposed at a relatively equal level.

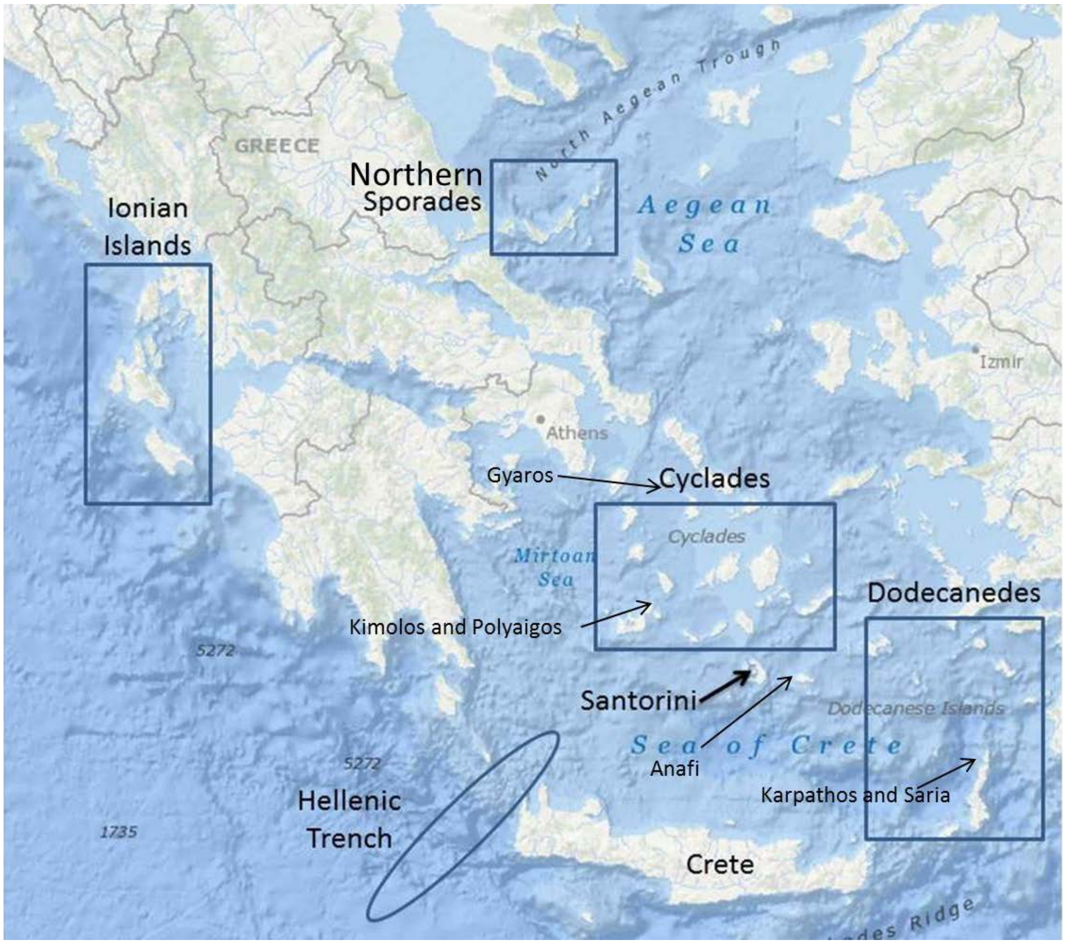
6.3.3.3 Exposure Estimates: Mediterranean Monk Seals

There is some demographic information on specific Mediterranean monk seal colonies in Greece; this information includes estimated numbers of individuals (Table 13). These colonies are spread throughout the eastern Mediterranean (Figure 9).

Table 13. Estimated number of individual Mediterranean monk seals in Greek colonies.

Island/Region	Estimated Number of Individuals	Citation
Gyaros	25	(Karamanlidis et al. 2013)
Northern Sporades	52	(MOm 2009)
Kimolos and Polyaigos	49	(MOm 2009)
North Karpathos and Saria	23	(MOm 2009)
Anafi	25	(MOm 2014b)
Ionian Islands	20	(Panou et al. 1993)

Figure 9. Map depicting locations of known Mediterranean monk seal colonies in Greece.



Due to their proximity to the proposed action area, we believe that it is most likely that individual Mediterranean monk seals from Anafi and the Kimolos and Polyaigos island complex could be exposed to the seismic activities. The island of Anafi is approximately 22 km from Santorini, and falls within the proposed action area. Because of this, we believe that all 25 individuals of all life stages on the Anafi colony could be exposed to the seismic activities. To the best of our knowledge, there has been no systematic survey of caves on Santorini to determine the number or presence of Mediterranean monk seals. However, based on recent stranding information (pers. comm. MOm, 2015), we believe it is possible that up to two Mediterranean monk seals could be present on Santorini island.

Kimolos and Polyaigos are approximately 80 and 70 km away from Santorini. Although both islands are physically outside the action area (i.e., the ensonified area of the Santorini tracklines), it is possible that individual monk seals from these colonies could travel into the ensonified area and be exposed to the proposed action.

As discussed earlier, there is little information available on the home ranges of Mediterranean monk seals, with the exception of a study conducted on 5 individuals in Greek waters (Adamantopoulou et al. 2011). The study group was comprised of one adult male, one adult female, one sub-adult male, and two sub-adult females. The adult female exhibited the longest straight-line distance (78 km), and had the largest home range (1,145 km^2); the adult male traveled a maximum straight-line distance of 30 km. The home ranges for the other sub-adults in the study were between 122 and 466 km^2 (Adamantopoulou et al. 2011).

The pupping season for Mediterranean monk seals in Greece lasts from August to December, and pups suckle for 3-4 months (MOm 2014b). Because the proposed action will occur during a time when we expect pups to be suckling in the caves, and thus not entering the action area, we believe that pups from Polyaigos and Kimolos are not likely to be exposed to the proposed seismic activities.

Based on the (Adamantopoulou et al. 2011) study, it does not appear likely that sub-adult or adult male Mediterranean monk seals would travel the 70-80 km from Polyaigos and Kimolos to the ensonified area and be exposed to the action. However, the available telemetry data indicate that adult female Mediterranean monk seals tend to travel farther than their cohorts. We thus believe that it is more likely that adult female Mediterranean monk seals could be exposed to the proposed seismic activities than other life stages or sexes because of the documented home range.

Determining the likelihood of monk seals from Polyaigos and Kimolos being present in the action area while the survey is occurring depends on other factors as well. Only a portion of the total tracklines would emit an ensonified area that would go near the islands of Polyaigos and Kimolos. Once those tracklines have been surveyed, the *Langseth* would continue to another area, away from Polyaigos and Kimolos, and outside the ranges of where we would expect monk seals from these colonies to be. Therefore, the number of days that monk seals from Polyaigos and Kimolos could potentially be exposed to the proposed seismic activities is fewer than the full 20 days of the survey.

Furthermore, we cannot know where specifically Mediterranean monk seals from Polyaigos and Kimolos will forage. There could be some oceanographic feature in the action area which is favorable for prey species that monk seals eat, and thus will be attracted to this area while foraging, but we cannot say for certain that monk seals from these colonies will be in this area during the survey. The northwest edge of the 160-dB ensonified area is approximately 60 km from the island complex. It is also possible that the monk seals would forage at another location and not be exposed to the action.

Based on the information before us, we do not believe it is likely that every individual of the Polyaigos and Kimolos colonies would be exposed to the proposed action. However, we do believe it is likely that some proportion of the Polyaigos and Kimolos monk seal colonies would be exposed. In the Kimolos Island complex (including Polyaigos), the data indicate that 8 pups are born every year (Politikos and Tzanetis 2009). This would indicate that there are, at minimum, 8 adult females in this colony, since Mediterranean monk seals give birth to a single pup (MOm 2014b). We therefore believe that it is possible that 8 adult female Mediterranean monk seals from the Polyaigos and Kimolos islands could be exposed to the proposed seismic activities.

Overall, on Anafi and Santorini, we expect that Mediterranean monk seals of all age classes are likely to be exposed. We expect that adult female Mediterranean monk seals from the Polyaigos and Kimolos island complex are likely to be exposed. Individuals from that colony are likely to be out traveling and foraging.

We agree with the Permit and Conservation Division’s exposure estimate for Mediterranean monk seals, and believe it is likely that up to 35 individuals could be exposed to the proposed seismic activities.

6.3.3.4 Exposure Estimates: Summary

In conclusion, we concur with the NMFS Permits and Conservation Division’s exposure estimates, with the exception of humpback whales, for the reasons explained above. The results of the exposure analysis are in Table 14. The numbers generated by NMFS represent the estimated number of individuals exposed throughout the entire action area. NSF presented their take authorization request based on the action area outside the territorial seas of Greece.

Table 14. Comparisons of the proposed take estimates calculated by the NMFS Permits and Conservation Division, and the NSF for ESA-listed marine mammals.

Species	NSF Exposure Estimate	NMFS Permits and Conservation Division Exposure Estimate	NMFS Interagency Cooperation Division Exposure Estimate
Humpback Whale	116	60	3
Fin Whale	50	120	120
Sei Whale	N/A	1	1
Sperm Whale	4	40	40
Mediterranean Monk Seal	4	35	35

6.3.3.5 Exposure of ESA-listed marine mammals 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 listed species to sound above the 160 dB re 1 μPa_{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). As such, their frequencies will attenuate more rapidly than those from airgun sources. ESA-listed individuals would experience higher levels of airgun sound well before either multibeam echosounder or sub-bottom profiler sound of equal amplitude would reach them.

While airguns are not operational, marine mammal observers will remain on duty to collect sighting data. If listed whales closely approached the vessel, the *Langseth* would take evasive actions to avoid a ship-strike and simultaneously avoid exposure to very high source levels. Ship strike has already been ruled out as a discountable effect, and we also rule out high-level ensonification of listed whales (multibeam echosounder source level = 242 dB re 1 μPa_{rms}; sub-bottom profiler source level = 204 dB re 1 μPa_{rms}). Boebel et al. (2006) 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. An individual would require exposure to 250–1,000 consecutive pulses from a sub-bottom profiler to be at risk for a temporary threshold shift (TTS). To be susceptible to TTS, a marine mammal 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 ESA-listed marine mammals (fewer than those exposed to airguns) could experience low-level multibeam echosounder and/or sub-bottom profiler sound exposure. We are unable to quantify the level of exposure from these secondary sound sources, but do not expect any exposure at levels sufficient to cause more than behavioral responses in some species capable of hearing frequencies produced by these systems. As discussed earlier, the sound levels produced by the airgun array are of primary concern in terms of exposure, due to their greater power.

6.3.4 Sea Turtles

NSF did not provide estimates of the expected number of ESA-listed turtles exposed to received levels ≥166 dB re 1 μPa_{rms}. Exposure estimates stem from the best available information on turtle densities and a planned ensonified area of approximately 12.6 km^2 along survey track lines, including areas of repeated exposure from adjacent track lines and turning legs. Exposures were developed by multiplying the ensonified area by the expected density. Based upon information presented in the *Response analysis*, we expect all exposures at the 166 dB re 1 μPa_{rms} level and above to constitute "take."

6.3.4.1 Exposure of ESA-listed turtles to airguns

NSF presented estimated distances for the 166 dB re 1 μPa_{rms} sound levels generated by the 36 45 in^3 GI-guns. When the array is towed at 12m, in deep water (>1000 m), the predicted established distance at received levels is 4,411 m; in shallower water (<100 m), this distance is 12,630 m. These are the distances at which sea turtles could experience fitness consequences as a result of the sound created by the airguns.

As discussed in the *Status of listed resources* section, there are three ESA-listed sea turtle species that are likely to be affected by the proposed action: green, leatherback, and loggerhead Mediterranean DPS sea turtles.

Estimating exposure for sea turtles in the action area was challenging, as there is scant information on sea turtle density or population estimates specific to Greek waters. To estimate exposure for sea turtles, we relied on recent status reviews, reports and scientific literature focusing on sea turtles in Greek and Mediterranean waters.

Of the three sea turtle species present in the Mediterranean, loggerheads are the most abundant, with 2,280-2,787 females nesting annually, compared to 339-360 green turtles (Broderick et al. 2002b). More recent estimates of green Mediterranean DPS turtles adult females are between 404 and 992 (Seminoff et al. 2015). There are no known leatherback sea turtle nesting sites in the Mediterranean (Caminas 1998). Thus, we believe that loggerheads are comparatively more likely to be exposed to the proposed action than the other sea turtle species. While there is some information available regarding numbers of sea turtle nesting sites and nesting females in the Mediterranean, there are no known population abundance or density estimates.

Loggerheads are known to nest in Greece, with areas of concentrated nesting occurring on the western coast of Peloponneus (i.e., on the Ionian Sea) and on the island of Rhodes in the east. Areas of sparse nesting occur elsewhere in the country, including on the Cyclades islands (i.e., within the action area) (Margaritoulis et al. 1995). Loggerhead nesting in Greece occurs from late May to mid-August (Broderick et al. 2002b), with hatchlings emerging 45-55 days later and residing in the ocean for several years (Bjorndal et al. 2000b). Hatchlings originating from Greek and Turkish nesting sites disperse throughout the Mediterranean (Casale and Mariani 2014), and could possibly be found in the action area, exposing them to the proposed seismic activities.

Juvenile and adult loggerheads of both sexes could also be exposed to the proposed seismic activities because at these life stages, individuals are typically foraging in the nearshore environment. The seismic survey tracklines go into shallow (<100m deep), nearshore waters. There are several known foraging areas throughout the Aegean Sea, including waters near Athens, Crete, Rhodes, and Kos (an island approximately 87 miles from Santorini) (Margaritoulis and Panagopoulou 2010 in (Casale and Margaritoulis 2010). A female loggerhead was telemetry tagged and released from northern Crete, and traveled around the Cyclades, including Santorini (Margaritoulis and Rees 2011).

Green turtles of all life stages and both sexes could be present in the action area during the proposed seismic activities, as they are considered year-round residents in the eastern Mediterranean, especially near Cyprus and Turkey (Broderick et al. 2002b). Major nesting sites for green turtles in the Mediterranean are found in Turkey and Cyprus, with hatchlings dispersing throughout the Mediterranean (Casale and Mariani 2014). Green turtle strandings have been reported in Greece at Rhodes island, approximately 150 miles from Santorini, with 16 strandings occurring in winter months from 1984-2011(Corsini-Foka et al. 2013). Other reports also record green turtle strandings in Greece, but mostly in the summer, and none in the Cyclades island region, near the action area (Panagopoulos et al. 2003b).

Leatherback sea turtles are present in the Mediterranean year-round, although there is no evidence to support regular nesting in the region (Caminas 1998; Groombridge 1990). We therefore do not expect leatherback hatchlings to be exposed to the proposed seismic activities. A comprehensive review of leatherback sighting and stranding records from 1981-2000 demonstrated that of the 411 leatherbacks sighted in the region, the majority were in the western Mediterranean, but several sightings were reported in Greek waters near the action area (Casale et al. 2003). Based on the carapace lengths of the leatherbacks recorded in that study, we believe that juvenile and adult leatherbacks of both sexes could potentially be exposed to the proposed seismic activities.

We are unable to quantify the level of sea turtle exposure. However, we do not expect sea turtle exposure to occur at high levels because the available information indicates that sea turtles are not reported to be highly abundant in the action area. As discussed earlier, there are no reliable sea turtle population estimates for Greek waters. Thus, it is not possible to quantify the proportion of the overall population that may be exposed to the proposed activity.

6.3.4.2 Exposure of ESA-listed turtles to multibeam echosounder and sub-bottom profiler

Sea turtles hear in the low frequency range. The multibeam echosounder and the SBP operate at 10.5-13 kHz, which emit sounds outside the hearing frequency of sea turtles. Thus, sea turtles are not expected to respond to sounds emitted by multibeam echosounder or sub-bottom profiler.

6.3.5 Response Analysis

A pulse of seismic airgun sound displaces water around the airgun and creates a wave of pressure, resulting in physical effects on the marine environment that can then affect marine organisms, such as listed whales, pinnipeds, and sea turtles considered in this Opinion. Possible responses considered in this analysis consist of:

- hearing 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, pinnipeds, and sea turtles in the action area.

6.3.5.1 Potential Response of Marine Mammals to Acoustic Sources

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. Threshold shift depends upon the duration, frequency, sound pressure, and rise time of the sound. A temporary threshold shift (TTS) results in a temporary hearing change (Finneran and Schlundt 2013). TTSs can last minutes to days. Full recovery is expected. 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, particularly in frequency ranges where animals are more sensitive, permanent threshold shift (PTS) can occur, meaning lost 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 generally specific to the frequencies over which exposure occurs but can extend to a half-octave above or below the center frequency of the source in tonal exposures (less evident in broadband noise such as the sound sources associated with the proposed action) (Kastak et al. 2005; Ketten 2012; Schlundt et al. 2000).

Few data are available to precisely define each listed species' hearing range, let alone its sensitivity and levels necessary to induce TTS or PTS. For pinnipeds in water, data are limited to measurements of TTS in harbor seals (*Phoca vitulina*), an elephant seal (*Mirounga angustirostris*), and California sea lions (*Zalophus californianus*) (Kastak et al. 199, 2005; Kastelein et al. 2012 b). The functional hearing frequency range for pinnipeds in air is 75 Hz to 30 kHz, and from 75 Hz to 75 kHz (Southall et al. 2007a). Baleen whales (e.g., sei, fin, and humpback) have an estimated functional hearing frequency range of 7 Hz to 22 kHz (Southall et al. 2007a).Based upon captive studies of odontocetes, our understanding of terrestrial mammal hearing, and extensive modeling, the best available information supports the position that sound levels at a given frequency would need to be ~186 dB SEL or ~196-201 dB re 1 μPa_{rms} in order to produce a low-level TTS from a single pulse (Southall et al. 2007c). 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. 2007c). 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 μPa_{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 μPa_{rms} isopleth and risk a TTS. Estimates that are conservative for species impact evaluation 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 or pinniped as a result of airgun exposure for several reasons. We expect that individuals will move away from the airgun array as it approaches. As the survey proceeds along each transect line and approaches ESA-listed individuals, the sound intensity increases, individuals will experience conditions (stress, loss of prey, discomfort, etc.) that prompt them to move away from the vessel and sound source and thus avoid exposures that would induce TTS or PTS. Ramp-ups would also reduce the probability of TTS-inducing exposure at the start of seismic surveys for the same reasons, as acoustic intensity increases, animals will move away. Furthermore, mitigation measures would be in place to initiate a power-down if individuals enter or are about to enter the 180 dB or 190 dB isopleth during full airgun operations, which is below the levels believed to be necessary for potential TTS. As stated in the *Exposure analysis*, each individual is expected to be potentially be exposed dozens of times to 160 dB re 1 μPa_{rms} levels. We do not expect this to produce a cumulative TTS, PTS, or other injury for several reasons. We expect that individuals will recover between each of these exposures, we expect monitoring to produce some degree of mitigation such that exposures will be reduced, and (as stated above), we expect individuals to generally move away at least a short distance as received sound levels increase, reducing the likelihood of exposure that is biologically meaningful.

Marine mammals and auditory interference (masking). Interference, or masking, occurs when a sound is a similar frequency and similar to or louder than the sound an animal is trying to hear (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 (Richardson et al. 1995). 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 sounds and vocalizations of ESA-listed whales, particularly baleen whales but also sperm whales. The proposed seismic surveys could mask whale calls at some of the lower frequencies. 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). Any masking that might occur would likely be temporary because seismic sources are not continuous and the seismic vessel would continue to transit through the area.

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 somewhat greater risk of 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.

The sound localization abilities of marine mammals suggest that, if signal and sound come from different directions, masking would not be as severe as the usual types of masking studies might suggest (Richardson et al., 1995). The dominant background noise may be highly directional if it comes from a particular anthropogenic source such as a ship or industrial site. Directional hearing may significantly reduce the masking effects of these sounds by improving the effective signal-to-sound ratio. In the cases of higher frequency hearing by the bottlenose dolphin, beluga whale, and killer whale, empirical evidence confirms that masking depends strongly on the relative directions of arrival of sound signals and the masking sound (Penner et al., 1986; Dubrovskiy, 1990; Bain et al., 1993; Bain and Dahlheim, 1994). Toothed whales and probably other marine mammals as well, have additional capabilities besides directional hearing that can facilitate detection of sounds in the presence of background sound. There is evidence that some toothed whales can shift the dominant frequencies of their echolocation signals from a frequency range with a lot of ambient sound toward frequencies with less noise (Au et al., 1974, 1985; Moore and Pawloski, 1990; Thomas and Turl, 1990; Romanenko and Kitain, 1992; Lesage et al., 1999). A few marine mammal species increase the source levels or alter the frequency of their calls in the presence of elevated sound levels (Dahlheim, 1987; Au, 1993; Lesage et al., 1993, 1999; Terhune, 1999; Foote et al., 2004; Parks et al., 2007, 2009; Di Iorio and Clark, 2010; Holt et al., 2009).

These data demonstrating adaptations for reduced masking pertain mainly to the very high frequency echolocation signals of toothed whales. There is less information about the existence of corresponding mechanisms at moderate or low frequencies or in other types of marine mammals. For example, Zaitseva et al. (1980) found that, for the bottlenose dolphin, the angular separation between a sound source and a masking noise source had little effect on the degree of masking when the sound frequency was 18 kHz, in contrast to the pronounced effect at higher frequencies. Studies have noted directional hearing at frequencies as low as 0.5-2 kHz in several marine mammals, including killer whales (Richardson et al., 1995a). This ability may be useful in reducing masking at these frequencies. In summary, high levels of sound generated by anthropogenic activities may act to mask the detection of weaker biologically important sounds by some marine mammals. This masking may be more prominent for lower frequencies. For higher frequencies, such as that used in echolocation by toothed whales, several mechanisms are available that may allow them to reduce the effects of such as that used in echolocation by toothed whales, several mechanisms are available that may allow them to reduce the effects of such masking.

Marine mammals and behavioral responses. We expect the greatest response to airgun sounds in terms of 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 can equate to take but are unlikely to be significant at the population level. 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 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). Dunn and Hernandez (2009) tracked blue whales during a seismic survey on the R/V *Maurice Ewing* (*Ewing*) in 2007 and did not observe changes in call rates and found no evidence of anomalous behavior that they could directly ascribe to the use of airguns at sound levels of approximately less than 145 dB re 1 μPa (Wilcock et al. 2014). 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 μ$\text{Pa}_{\text{p-p}}$ (Madsen et al. 2002; McCall Howard 1999). Some exposed individuals may cease calling in response to the *Langseth's* airguns. If individuals ceased calling in response to the *Langseth's* airguns during the course of the proposed survey, the effect would likely be temporary.

There are numerous studies of the responses of some baleen whale to airguns. Although responses to lower-amplitude sounds are known, most studies seem to support a threshold of ~160 dB re 1 μPa_{rms} as the received sound level to cause behavioral responses other than vocalization changes (Richardson et al. 1995c). Activity of individuals seems to influence response (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. 1999). 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 μPa_{rms} exposures at distances of 20-30 km, but only changed dive and respiratory patterns while feeding and showed avoidance at higher received sound levels (152–178 dB re 1 μPa_{rms}) (Harris et al. 2007; Ljungblad et al. 1988; Miller et al. 1999; Miller et al. 2005; Richardson et al. 1995c; Richardson et al. 1999; 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). 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, appear to reoccupy areas within a day (Richardson et al. 1986). We do not know whether the individuals exposed in these ensonified areas are the same returning or whether individuals that tolerate repeat exposures may still experience a stress response.

Gray whales respond similarly. Gray whales discontinued feeding and/or moved away at received sound levels of 163 dB re 1 μPa_{rms} (Bain and Williams 2006; Gailey et al. 2007; Johnson et al. 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 μPa_{rms} (Malme and Miles 1985; Malme et al. 1984). As with bowheads, habitat continues to be used despite frequent seismic survey activity, but long-term effects have not been identified, if they are present at all (Malme et al. 1984). Johnson et al. (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 exhibit 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 μPa_{rms} when females with calves were present, or 7-12 km from the seismic source (McCauley et al. 2000a; McCauley et al. 1998). A startle response occurred as low as 112 dB re 1 μPa_{rms}. Closest approaches were generally limited to 3-4 km, although some individuals (mainly males) approached to within 100 m on occasion where sound levels were 179 dB re 1 μPa_{rms}. Changes in course and speed generally occurred at estimated received level of 157–164 dB re 1 μPa_{rms}.

Natural sources of sound also influence humpback behavior. Migrating humpbacks showed evidence of a Lombard effect in Australia, increasing vocalization in response to wind-dependent background noise (Dunlop et al. 2014a). Since natural sources of noise alone can influence whale behavior, additional anthropogenic sources could also add to these effects.

Multiple factors may contribute to the degree of response exhibited by migrating humpbacks. In a preliminary study examining the responses by migrating humpbacks of exposure to a $20in^3$ air gun, researchers found that the whales' behavior seemed to be influenced by social effects; "whale groups decreased dive time slightly and decreased speed towards the source, but there were similar responses to the control" (*i.e.*, a towed air gun, not in operation) (Dunlop et al. 2014b). Whales in groups may pick up on responses by other individuals in the group and react. The results of this continued study are still pending, and will examine the effects of a full size commercial air gun array on humpback behavior (Dunlop et al. 2014b).

Feeding humpbacks appear to be somewhat more tolerant. Humpback whales along Alaska startled at 150–169 dB re 1 μPa and no clear evidence of avoidance was apparent at received levels up to 172 re 1 μPa_{rms} (Malme et al. 1984; Malme et al. 1985). Potter et al. (2007) found that humpbacks on feeding grounds in the Atlantic did exhibit localized avoidance to airguns. Among humpback whales on Angolan breeding grounds, no clear difference was observed in encounter rate or point of closest approach during seismic versus non-seismic periods (Weir 2008).

Observational data are sparse for specific baleen whale life 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 μPa_{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. 2000b; Madsen et al. 2006; Miller et al. 2009; Moulton et al. 2006a; Moulton and Miller 2005; Stone 2003; Stone and Tasker 2006; Weir 2008). Detailed study of Gulf of Mexico sperm whales suggests some alteration in foraging from <130-162 dB re 1 μPa_{p-p}, although other behavioral reactions were not noted by several authors (Gordon et al. 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 during the course of seismic surveys (Goold and Fish 1998), so even though this species generally hears at higher frequencies, this does not mean that it cannot hear airgun sounds. 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 breeding encounters toothed whales sometimes are extremely tolerant of noise pulses (NMFS 2006b).

For whales exposed to seismic airguns during the proposed activities, behavioral changes stemming from airgun exposure may result in loss of feeding opportunities. We expect listed whales exposed to seismic airgun sound will exhibit an avoidance reaction, displacing individuals from the area 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 whales would be displaced from that is essential for breeding if 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.

Pinnipeds are not likely to show a strong avoidance reaction to the airgun sources proposed for use. Visual monitoring from seismic vessels has shown only slight (if any) avoidance of airguns by pinnipeds and only slight (if any) changes in behavior. Monitoring work in the Alaskan Beaufort Sea during 1996-2001 provided considerable information regarding the behavior of Arctic ice seals exposed to seismic pulses (Harris et al. 2001; Moulton and Lawson 2002). These seismic projects usually involved arrays of 6 to 16 airguns with total volumes of 560 to 1,500 in^3. The combined results suggest that some seals avoid the immediate area around seismic vessels. In most survey years, ringed seal (*Phoca hispida*) sightings tended to be farther away from the seismic vessel when the airguns were operating than when they were not (Moulton and Lawson 2002). However, these avoidance movements were relatively small, on the order of 100 m (328 ft) to a few hundreds of meters, and many seals remained within 100-200 m (328-656 ft) of the trackline as the operating airgun array passed by the animals. Seal sighting rates at the water surface were lower during airgun array operations than during no-airgun periods in each survey year except 1997. Similarly, seals are often very tolerant of pulsed sounds from seal-scaring devices (Mate and Harvey 1987; Jefferson and Curry 1994; Richardson et al. 1995). However, initial telemetry work suggests that avoidance and other behavioral reactions by two other species of seals to small airgun sources may at times be stronger than evident to date from visual studies of pinniped reactions to airguns (Thompson et al. 1998).

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. Other effects like neurological effects, bubble formation, and other types of organ or tissue damage could occur, but similar to stress, these effects are not readily observable.

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 2011). Romano et al. (2004) found beluga whales and bottlenose dolphins exposed to a seismic water gun (up to 228 dB re 1 μPa · m_{p-p}) and single pure tones (up to 201 dB re 1 μPa) had increases in stress chemicals, including catecholamines, which could affect an individual’s ability to fight off disease. During the time following September 11, 2001, shipping traffic and associated ocean noise decreased along the northeastern U.S.; this decrease in ocean noise was associated with a significant decline in fecal stress hormones in North Atlantic right whales, providing evidence that chronic exposure to increased noise levels, although not acutely injurious, can produce stress (Rolland et al. 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 listed individual to experience a stress response at high levels. We assume that a stress response could be associated with displacement or, if individuals remain in a stressful environment, the stressor (sounds associated with the airgun, multibeam echosounder, or sub-bottom profiler) will dissipate in a short period as the vessel (and stressors) transects away without significant or long-term harm to the individual via the stress response.

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

It is possible that an animal's prior exposure to seismic sounds influences its future response. We have little information available to us as to what response individuals would have to future exposures to seismic sources compared to prior experience. If prior exposure produces a learned response, then this subsequent learned 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*).

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

Suggestions that there was a link between seismic surveys and strandings of humpback whales in Brazil (Engel et al., 2004) were not well founded (IAGC, 2004; IWC, 2007). In September 2002, two Cuvier's beaked whales (*Ziphius cavirostris*) stranded in the Gulf of California, Mexico. The *R/V Maurice 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). Numerous studies suggest that the physiology, behavior, habitat relationships, age, or condition of cetaceans may cause them to strand or might pre-dispose them to strand when exposed to another phenomenon. These suggestions are consistent with the conclusions of numerous other studies that have demonstrated that combinations of dissimilar stressors commonly combine to kill an animal or dramatically reduce its fitness, even though one exposure without the other does not produce the same result (Chroussos, 2000; Creel, 2005; DeVries et al., 2003; Fair and Becker, 2000; Foley et al., 2001; Moberg, 2000; Relyea, 2005a; 2005b, Romero, 2004; Sih et al., 2004). At present, the factors of seismic airguns that may contribute to marine mammal strandings are unknown and we have no evidence to lead us to believe that aspects of the airgun array proposed to for use will cause marine mammal strandings. We do not expect listed whales and pinnipeds 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 listed whales and pinipeds. Unfortunately, species-specific information on the prey of listed whales and pinnipeds 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. 1996b; McCauley et al. 2000a; McCauley et al. 2000b; McCauley et al. 2003; Popper et al. 2005; Santulli et al. 1999). Lethal effects, if any, are expected within a few meters of the airgun array (Buchanan et al. 2004; Dalen and Knutsen 1986). We expect fish to be capable of moving away from the airgun array if it causes them discomfort.

More evidence exists for sub-lethal effects. Several species at various life stages have been exposed to high-intensity sound sources (220-242 dB re 1 μPa) at close distances, with some cases of injury (Booman et al. 1996; McCauley et al. 2003). TTS was not found in whitefish at received levels of ~175 dB re 1 $\mu Pa^2 \cdot s$, but pike did show 10-15 dB of hearing loss with recovery within 1 day (Popper et al. 2005). Caged pink snapper have experienced PTS when exposed over 600 times to received seismic sound levels of 165-209 dB re 1 μPa_{p-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 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 μPa_{0-p} and alarm responses at >177 dB re 1 μPa_{0-p} (Pearson et al. 1992). Fish also tightened schools and shifted their distribution downward. Normal position and behavior resumed 20-60 minutes after seismic firing ceased. A downward shift was also noted by Skalski et al. (1992) at received seismic sounds of 186–191 re 1 μPa_{0-p}. Caged European sea bass showed elevated stress levels when exposed to airguns, but levels returned to normal after 3 days (Skalski et al. 1992). These fish also showed a startle response when the survey vessel was as much as 2.5 km away; this response increased in severity as the vessel approached and sound levels increased, but returned to normal after 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. 1996b). Lesser sandeels exhibited initial startle responses and upward vertical movements before fleeing from the survey area upon approach of an active seismic vessel (Hassel et al. 2003; Hassel et al. 2004). McCauley et al. (2000; 2000a) found smaller fish show startle responses at lower levels than larger fish in a variety of fish species and generally observed responses at received sound levels of 156–161 dB re 1 μPa_{rms}, but responses tended to decrease over time suggesting habituation. As with previous studies, caged fish showed increases in swimming speeds and downward vertical shifts. Pollock did not respond to airgun sounds received at 195–218 dB re 1 $\mu Pa_{0\text{-}p}$, but did exhibit continual startle responses and fled from the seismic source when visible (Wardle et al. 2001). Blue whiting and mesopelagic fishes were found to redistribute 20–50 m deeper in response to airgun ensonification and a shift away from the survey area was also found (Slotte et al. 2004). 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. (1996a) found no differences in trawl catch data before and after seismic operations and echosurveys of fish occurrence did not reveal differences in pelagic biomass. However, fish kept in cages did show behavioral responses to approaching airguns.

Squid responses to airguns have also been studied, although to a lesser extent than fishes. In response to airgun exposure, squid exhibited both startle and avoidance responses at received sound levels of 174 dB re 1 μPa_{rms} by first ejecting ink and then moving rapidly away from the area (McCauley 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 μPa_{rms}. Tenera Environmental (2011) reported that Norris and Mohl (1983, summarized in Mariyasu et al. 2004) observed lethal effects in squid (*Loligo vulgaris*) at levels of 246 to 252 dB after 3 to 11 minutes. Andre et al. (2011) exposed four cephalopod species (*Loligo vulgaris*, *Sepia officinalis*, *Octopus vulgaris*, and *Ilex coindetii*) to two hours of continuous sound from 50-400 Hz at 157 +/- 5 dB re

1 µPa. They reported lesions to the sensory hair cells of the statocysts of the exposed animals that increased in severity with time, suggesting that cephalopods are particularly sensitive to low-frequency sound. The received sound pressure level was 157 +/- 5 dB re 1 µPa, with peak levels at 175 dB re 1 µPa. 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. Another laboratory story observed abnormalities in larval scallops after exposure to low frequency noise in tanks (de Soto et al. 2013). 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 listed baleen whales) to experience effects from airgun sound. Although humpback whales consume fish regularly, we expect that any disruption to their prey will be temporary, if at all. Therefore, we do not expect any adverse effects from lack of prey availability to baleen whales. Sperm whales regularly feed on squid and some fishes and we expect individuals to feed while in the action area during the proposed survey. Based upon the best available information, fishes and squids ensonified by the ~160 dB isopleths could vacate the area and/or dive to greater depths, and be more alert for predators. We do not expect indirect effects from airgun activities through reduced feeding opportunities for listed whales and pinnipeds 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 listed whales and pinnipeds 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 marine mammals except humpback and sperm whales. We expect that these systems will produce harmonic components in a frequency range above and below the center 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 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 or sei whales are exposed, they are unlikely to hear these frequencies well (if at all) and a response is not expected.

Assumptions for humpback and sperm whale hearing are much different than for other listed whales. Humpback and sperm whales vocalize between 3.5-12.6 kHz and an audiogram of a juvenile sperm whale provides direct support for hearing over this entire range (Au 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 1986b; 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. The animal would have to pass the transducer at close range and be swimming at speeds similar to the vessel in order to receive the multiple pulses that might result in sufficient exposure to casue TTS. 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 not while traveling (Andre and Jurado 1997; André et al. 1997).

Investigations stemming from a 2008 stranding event in Madagascar indicated 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 (*Peponocephala electra*) (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. 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).

Stranding events associated with the operation of naval sonar suggest that mid-frequency sonar sounds may have the capacity to cause serious impacts to marine mammals. The sonars proposed for use by L-DEO differ from sonars used during naval operations, which generally have a longer pulse duration and more horizontal orientation than the more downward-directed multibeam echosounder and sub-bottom profiler. The sound energy received by any individuals exposed to the multibeam echosounder and sub-bottom profiler sources during the proposed activities is lower relative to naval sonars, as is the duration of exposure. The area of possible influence for the multibeam echosounder and sub-bottom profiler is also much smaller, consisting of a narrow zone close to and below the source vessel. Although navigational sonars are operated routinely by thousands of vessels around the world, strandings have been correlated to use of these sonars. 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 appreciably occur due to multibeam echosounder or sub-bottom profiler signal directionality, low duty cycle, and the brief period when an individual could be within its beam. 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. Behavioral responses to the mulitbeam echosounder and sub-bottom profiler are likely to be similar to the other pulsed sources discussed earlier if received at the same levels. However, the pulsed signals from the sub-bottom profiler are considerably weaker than those from the multibeam echosounder. Also, we do not expect hearing impairment and other physical effects if the animal is in the area, and it would have to pass the transducers at close range and in order to be subjected to sound levels that could cause temporary threshold shift.

6.3.5.2 Potential Response of Sea turtles to Acoustic Sources

As with marine mammals, sea turtles may experience

- hearing 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, because of the mitigation measures and our expectation that turtles would move away from sounds from the airgun array, we do not expect turtles to be exposed to sound levels that would result in TTS or PTS.

Sea turtles and behavioral responses. As with ESA-listed marine mammals, it is likely that sea turtles will experience behavioral responses in the form of avoidance. O'Hara and Wilcox (1990) found loggerhead sea turtles exhibited an avoidance reaction at an estimated sound level of 175–176 dB re 1 µPa $_{\text{rms}}$ (or slightly less) in a shallow canal. 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 1994a; 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, listed sea turtles will not hear these sounds even if they are exposed and are not expected to respond to them.

6.4 Cumulative Effects

"Cumulative effects" are those effects of future state or private activities, not involving Federal activities, that are reasonably certain to occur within the action area of the Federal action subject to consultation (50 CFR 402.02). Future Federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the ESA.

6.5 Integration and Synthesis

The Integration and Synthesis section is the final step in our assessment of the risk posed to species and critical habitat as a result of implementing the proposed action. In this section, we add the effects of the action (Section 6) to the environmental baseline (Section 5) and the cumulative effects (Section 6.4) to formulate the agency's biological opinion as to whether the proposed action is likely to: (1) reduce appreciably the likelihood of both the survival and recovery of a ESA-listed species in the wild by reducing its numbers, reproduction, or distribution; or (2) reduce the value of designated or proposed critical habitat for the conservation of the species. These assessments are made in full consideration of the status of the species and critical habitat (Section 4).

The following discussions separately summarize the probable risks the proposed action poses to threatened and endangered species and critical habitat that are likely to be exposed. These summaries integrate the exposure profiles presented previously with the results of our response analyses for each of the actions considered in this opinion.

ESA-Listed marine mammals. The NSF proposes to allow the use of its vessel, the *Langseth*, to conduct a seismic survey by L-DEO that could incidentally harass several ESA-listed marine mammal species; and PR1 proposes to authorize the incidental take of marine mammals. These species include: Mediterranean monk seals, fin whales, humpback whales, sei whales, and sperm whales, all of which 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. 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 Mediterranean and Atlantic Ocean 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 marine mammals by proposed seismic activities. As discussed in the exposure analysis, we expect up to 35 Mediterranean monk seals, 120 fin, 1 sei, 3 humpback, and 40 sperm whales could be exposed to airgun sounds which will elicit a behavioral response of temporarily moving out of the area. We expect a low-level, transitory stress response to accompany this behavior. The number of individuals exposed is a small fraction of the populations, with some individual re-exposure and reactions. These exposures should not limit the fitness of any single individual. The other actions we considered in the Opinion, the operation of multibeam echosounder and sub-bottom profiler systems, are not expected to be audible to fin or sei whales and consequently are not expected to have any direct effects on these species. However, humpback, and sperm whales could hear sounds produced by these systems. Responses could include cessation of vocalization by sperm whales and/or movement out of the survey area by 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 or reoccupy the habitat from which they were displaced within a period of days (or less). We do not expect these effects to have fitness consequences for any individual. The *Effects Analysis* also found that, although sperm whales may experience temporarily reduced feeding opportunities; this indirect effect would be transient and not reduce individual fitness of any whale. Based upon these findings, the risk of fitness consequences to any single individual is not expected to translate to population or species-level consequences. Overall, we do not expect a fitness reduction to any individual whale from the survey or IHA. As such, we do not expect fitness consequences to populations or listed whale species as a whole.

ESA-Listed turtles. ESA-listed turtles that are expected to occur within the action area include green 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 green, leatherback, and loggerhead sea turtles could experience exposure to airgun sounds and be harassed by these sounds. These sounds may induce a temporary increase 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 very small proportion of each population would be affected. We expect transient responses that do not affect the fitness of any one individual. We do not expect impairment of local nesting by the proposed survey. As we do not expect any sea turtle to be capable of hearing signals produced by the multibeam echosounder and sub-bottom profiler systems, we do not expect direct effects from these systems on sea turtle fitness. We do not anticipate any indirect effects from the proposed actions to influence sea turtles. Overall, we do not expect any individual sea turtle to undergo a fitness consequence. Based upon these findings, the risk of fitness consequences to any single individual is not expected to translate to population or species-level consequences. Because we do not expect individual sea turtles to experience fitness reductions, we also do not expect reductions in the viability of the populations these individuals belong or the viability of the species those populations comprise.

7 CONCLUSION

After reviewing the current status of the ESA-listed species, the environmental baseline within the action area, the effects of the proposed action, any effects of interrelated and interdependent actions, and cumulative effects, it is NMFS' biological opinion that the proposed action is not likely to jeopardize the continued existence of Mediterranean monk seal, fin, sei, humpback, or sperm whales, or green, leatherback or loggerhead sea turtles. No critical habitat has been designated or proposed for this species; therefore, none will be affected.

8 INCIDENTAL TAKE STATEMENT

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

8.1 Amount or Extent of Take

Section 7 regulations require NMFS to specify the impact of any incidental take of endangered or threatened species; that is, the amount or extent, of such incidental taking on the species (50 CFR § 402.14(i)(1)(i)). The amount of take represents the number of individuals that are expected to be taken by actions while the extent of take or "the extent of land or marine area that may be affected by an action" may be used if we cannot assign numerical limits for animals that could be incidentally taken during the course of an action (51 FR 19953).

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 Mediterranean monk seals, fin, sei, humpback, and sperm whales pursuant to section 101(a)(5)(D) of the MMPA. The final authorization would 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 would be 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.

As discussed previously, we have jurisdiction to authorize incidental take of ESA-listed species in areas outside the territorial seas of Greece (i.e., >6 nautical miles). Earlier, we examined the probable exposure of ESA-listed species in the full extent of the action area (6.3.3.4) to make our jeopardy determination. Here, we describe the amount of incidental take authorized for the action area outside the territorial seas of Greece.

Fin whales commonly occur offshore, but also can be found in coastal areas (Aguilar 2009). Sperm whale density in the Mediterranean is highest along the 1,000m contour, with distance from the coast ranging from 2.7-29.4 km (Frantzis et al. 2014). Since both of these species can also occur closer to shore, so we recognize that it is possible that fin and sperm whales could be exposed inside and outside the 6 n mi boundary. For fin and sperm whales, the amount of daily ensonified area outside the 6 n mi boundary was determined to be 1,211 km^2. This figure was multiplied by the winter density estimates developed by Laran et al. (2010). Based on the available population information for these species, and what we know about their behavior, we believe that the calculation results in a reasonable estimate of the amount of fin and sperm whales exposed outside the territorial seas of Greece. We are authorizing the same numbers in the ITS for these species as in the IHA (**Table 15**).

Because there were no density estimates available for Mediterranean monk seals, sei or humpback whales, we were not able to partition the amount of take that we expected to occur outside the territorial seas of Greece. However, based on what we know about the biology of these species, we can speak generally about where we expect Mediterranean monk seals, sei and humpbacks whales will be with respect to the territorial seas boundary.

As discussed in section 6.3.3.3, Mediterranean monk seals are present in the action area, and are likely to be exposed to the seismic survey. The available information indicates that Mediterranean monks seals leave their caves and forage, with home ranges from 122 to 1,145 km^2 (Adamantopoulou et al. 2011). There are reports of monk seals stranding in areas without known local populations (as happened on Santorini; pers. comm. MOm, 2015) within Greece, and further throughout the Mediterranean in places where the monk seal has not been seen for decades (Alfaghi et al. 2013; Inanmaz et al. 2014; Scheinin et al. 2011a). We believe it is therefore possible that Mediterranean monk seals could be outside the territorial seas of Greece. If monk seals are exposed while outside the 6 n mi boundary, it is likely that it would occur while the monk seals are out foraging or traveling. We are not aware of any known foraging areas, or specific routes that monk seals travel, so are not able to make precise predictions about how many Mediterranean monk seals will be inside or outside the 6 n mi boundary during the seismic survey. As such, we are not able to partition a take estimate on a finer scale than our exposure estimate of 35 Mediterranean monk seals. Therefore, we will issue take of 35 Mediterranean monk seals in the ITS.

Sei whales are typically found in deep waters and along continental shelf edges (Hain et al. 1985a). Similar to humpbacks, sei whales migrate in winter (Perry et al. 1999b). Since the survey will occur in November and December, it is most probable that sei whales would be encountered while they are migrating. Based on their typical habitat, it is also most likely that they would be encountered in deep water further from shore, outside the territorial seas of Greece. Therefore, we believe it is appropriate to issue 1 take of sei whales in the ITS.

As discussed above, humpback whales are considered rare in the Mediterranean, with only 3 documented in Greece in the last hundred years (Frantzis et al. 2004b). Humpback whales migrate in winter, and these migrations occur through deep pelagic waters (Winn and Reichley 1985a). If present during the seismic survey, it is probable that humpbacks would be encountered while in transit over deeper water further from shore, and thus outside the 6 nautical mile boundary. Therefore, we believe it is appropriate to issue 3 takes of humpback whales in the ITS.

The NMFS anticipates the proposed seismic survey in the eastern Mediterranean is likely to result in the incidental take of ESA-listed species by harassment (Table 15). We expect up to 35 Mediterranean monk seals, 40 fin, 1 sei, 3 humpback, and 20 individual sperm whales could be exposed to airgun sounds during the course of the proposed seismic survey, which will elicit a behavioral response that would constitute harassment. Harassment is expected to occur at received levels above 160 dB re: 1 μPa for ESA-listed whales and Mediterranean monk seals. Additional exposures to the same individuals sufficient to elicit responses may also occur.

For all species of marine mammals, this incidental take would result 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.

Table 15. Amount of incidental take of ESA-listed marine mammals authorized by the Incidental Take Statement.

Species	Number of Individuals Authorized for Incidental Take
Fin whale	40
Sei whale	1
Humpback whale	3
Sperm whale	20
Mediterranean monk seal	35

We also expect individual green, leatherback, and loggerhead Mediterranean Sea DPS sea turtles could be exposed to airgun sounds during the course of the proposed seismic survey that will elicit a behavioral response that would constitute harassment. No death or injury is expected for individuals who are exposed to the seismic activities. Harassment for sea turtles is expected to occur at received levels above 166 dB re: 1 μPa, which includes a 12.6 km^2 area in the eastern Mediterranean based upon the propagation and trackline estimates provided by the NSF. If these amounts change, then incidental take for marine mammals or sea turtles may be exceeded. As such, if more trackline, greater estimates of sound propagation, and/or increases in airgun source levels occur, re-initiation of consultation may be necessary. As we cannot determine the number of individuals to which harassment will occur, we expect the extent of exposure will occur within the 166 dB isopleth of the *Langseth's* airgun array.

8.2 Effects of the Take

In this Opinion, NMFS determined that the amount or extent of anticipated take, coupled with other effects of the proposed action, is not likely to result in jeopardy to the species or destruction or adverse modification of critical habitat.

8.3 Reasonable and Prudent Measures

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

"Reasonable and prudent measures" are nondiscretionary measures to minimize the amount or extent of incidental take (50 CFR 402.02). NMFS believes the reasonable and prudent measures described below are necessary and appropriate to minimize the impacts of incidental take on threatened and endangered species:

- The Permits and Conservation Division and the NSF must ensure that the L-DEO implements and monitors the effectiveness of mitigation measures incorporated as part of the proposed authorization of the incidental taking of Mediterranean monk seals, fin, sei, humpback, and sperm whales pursuant to section 101(a)(5)(D) of the MMPA and as specified below for green, leatherback, and loggerhead sea turtles. In addition, the Permits and Conservation Division must ensure that the provisions of the IHA are carried out, and to inform the ESA Interagency Cooperation Division if take is exceeded.

8.4 Terms and Conditions

To be exempt from the prohibitions of section 9 of the ESA, the NSF, 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, 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.

This ITS does not cover incidental takes of marine mammals in the territorial seas of foreign nations, as the neither the ESA nor the MMPA do not apply in those waters. Take coverage is only for the area outside Greek territorial waters (i.e., outside of 6 nautical miles). The L-DEO is required to follow the requirements presented in the IHA.

To implement the reasonable and prudent measures, the L-DEO and the NMFS' Permits and Conservation Division shall ensure that:

a. The ITS limits the incidental taking of ESA-listed species by harassment only, to the species listed in Table 15.

    i. During the seismic activities, if the Holder of this ITS encounters any ESA-listed species that are not listed in the ITS for authorized taking and are likely to be exposed to sound pressure levels greater than or equal to 160 decibels (dB) re: 1 μPa, then the Holder of the ITS must alter speed or course or shut-down the airguns to avoid take.

b. In addition, The IHA prohibits the taking by serious injury or mortality of any of the species listed in Condition 3 of the IHA or the taking of any other kind of species of marine mammal. Thus, if this were to occur, it may result in the modification, suspension, or revocation of the IHA, and trigger a requirement for reinitiation under the ESA.

c. Both the action described in the opinion and the IHA limit the methods authorized for taking by harassment to the following acoustic sources without an amendment to the IHA or a request for reinitiation:
   i. An airgun array with a total capacity of 6,600 cubic inches (in^3) (or smaller).

d. Lamont-Doherty will not operate the multi-beam echosounder or the sub-bottom profiler during transit to or from the survey areas.

4. Reporting Prohibited Take

The Holder of the IHA must report the taking of any marine mammal in a manner prohibited under this Authorization immediately to the Chief, Permits and Conservation Division, Office of Protected Resources, National Marine Fisheries Service, at 301-427-8401 and/ or by email to Jolie.Harrison@noaa.gov and to Cathy Tortorici ESA Interagency Cooperation Division Chief at Cathy.Tortorici@noaa.gov.

5. Cooperation

We require the Holder of this Authorization to cooperate with the Office of Protected Resources, National Marine Fisheries Service, and any other Greek and U.S. Federal agency requirements for monitoring the impacts of the activity on marine mammals.

6. Mitigation and Monitoring Requirements

We require the Holder of this ITS to implement the following mitigation and monitoring requirements when conducting the specified activities to achieve the least practicable adverse impact on affected marine mammal species or stocks:

Visual Observers

a. Use two, National Marine Fisheries Service-qualified, vessel-based Protected Species Visual Observers (visual observers) to watch for and monitor marine mammals near the seismic source vessel during daytime airgun operations (from nautical twilight-dawn to nautical twilight-dusk) and before and during start-ups of airguns day or night.
   i. At least one visual observer will be on watch during meal times and restroom breaks.
   ii. Visual observer shifts will last no longer than four hours at a time.
   iii. Visual observers will also conduct monitoring while the *Langseth* crew deploy and recover the airgun array, ocean bottom seismometers, and hydrophone streamer from the water.
   iv. When feasible, visual observers will conduct observations during daytime periods when the seismic system is not operating for comparison of sighting rates and behavioral reactions during, between, and after airgun operations.
   v. The *Langseth's* vessel crew will also assist in detecting marine mammals, when practicable.
   vi. Visual observers will have access to reticle binoculars (7×50 Steiner), big-eye binoculars (25×150), optical range finders, and night vision devices.

Exclusion Zones

b. Establish a 180-decibel (dB) and 190-dB exclusion zone for cetaceans and pinnipeds, respectively, before starting the airgun array (6,600 in^3 or smaller); and establish a 180-dB and 190-dB exclusion zone for cetaceans and pinnipeds, respectively, for the single airgun (40 in^3). Observers will use the predicted radius distance for the 180-dB and 190-dB exclusion zones for mitigation shown in Table 2 (of the IHA).

Visual Monitoring at the Start of Airgun Operations

c. Monitor the entire extent of the exclusion zones for at least 30 minutes (day or night) prior to the ramp-up of airgun operations after a shutdown.

d. Delay airgun operations if the visual observer sees a cetacean within the 180-dB exclusion zone (as defined in Table 2 of the IHA) until the marine mammal(s) has left the area.

Delay airgun operations if the visual observer sees a pinniped within the 190-dB exclusion zone (as defined in Table 2 of the IHA) until the marine mammal(s) has left the area.

i. If the visual 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, he/she should assume that the animal has moved beyond the relevant exclusion zone (as defined in Table 2 of the IHA).

ii. If, for any reason the visual observer cannot see the full relevant exclusion zone (as defined in Table 2 of the IHA) for the entire 30 minutes (*i.e*., rough seas, fog, darkness), or if marine mammals are near, approaching, or within zone, the *Langseth* may not resume airgun operations.

iii. If one airgun is already running at a source level of at least 180 dB re: 1 μPa, the *Langseth* may start the second gun–and subsequent airguns–without observing relevant exclusion zones for 30 minutes, provided that the observers have not seen any marine mammals near the relevant exclusion zones (in accordance with Condition 6(b) of the IHA).

Passive Acoustic Monitoring

e. Utilize the passive acoustic monitoring (PAM) system, to the maximum extent practicable, to detect and allow some localization of marine mammals around the *Langseth* during all airgun operations and during most periods when airguns are not operating. One visual observer and/or bioacoustician will monitor the PAM at all times in shifts no longer than 6 hours. A bioacoustician shall design and set up the PAM system and be present to operate or oversee PAM, and available when technical issues occur during the survey.

f. Do and record the following when an observer detects an animal by the PAM:

i. Notify the visual observer immediately of a vocalizing marine mammal so a power-down or shut-down can be initiated, if required;

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

Ramp-Up Procedures

g. Implement a ‘‘ramp-up’’ procedure when starting the airguns at the beginning of seismic operations or any time after the entire array has shutdown, which means starting the smallest gun first and adding airguns in a sequence such that the source level of the array will increase in steps not exceeding approximately 6 dB per 5-minute period. During ramp-up, the observers will monitor the exclusion zones, and if the observers sight marine mammals, the *Langseth* will implement a course/speed alteration, power-down, or shutdown as though the full array were operational.

Recording Visual Detections

h. Visual observers must record the following information when they detect a marine mammal:

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

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

    iii. The data listed under 6(h)(ii) of the IHA 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.

Speed or Course Alteration

i. Alter speed or course during seismic operations if a marine mammal, based on its position and relative motion, appears likely to enter the relevant exclusion zone. If speed or course alteration is not safe or practicable, or if after alteration the marine mammal still appears likely to enter the relevant exclusion zone, Lamont-Doherty will implement further mitigation measures, such as a power-down or shutdown.

Power-Down Procedures

j. Power down the airguns if a visual observer detects a marine mammal within, approaching, or entering the relevant exclusion zone (as defined in Table 2 of the IHA). A power-down means reducing the number of operating airguns to a single operating 40 in^3 airgun. This would reduce the relevant exclusion zone to the degree that the animal(s) is/are outside of that zone. When appropriate or possible, power-down of the airgun array shall also occur when the vessel is moving from the end of one trackline to the start of the next trackline.

Resuming Airgun Operations after a Power-Down

k. Following a power-down, if the marine mammal approaches the smaller exclusion zone (as defined in Table 2 of the IHA), then the *Langseth* must completely shut down the airguns. Airgun activity will not resume until the observer has visually observed the marine mammal(s) exiting the exclusion zone and is not likely to return, or the observer has not seen the animal within the relevant exclusion zone for 15 minutes for species with shorter dive durations (*i.e*., small odontocetes) or 30 minutes for species with longer dive durations (*i.e*., mysticetes and large odontocetes, including sperm, pygmy sperm, dwarf sperm, and beaked whales) and pinnipeds.

l. Following a power-down and subsequent animal departure, the *Langseth* may resume airgun operations at full power. Initiation requires that the observers can effectively monitor the full exclusion zones described in Condition 6(b) of the IHA. If the observer sees a marine mammal within or about to enter the relevant zones then the *Langseth* will implement a course/speed alteration, power-down, or shutdown.

Shutdown Procedures

m. Shutdown the airgun(s) if a visual observer detects a marine mammal within, approaching, or entering the relevant exclusion zone (as defined in Table 2 of the IHA). A shutdown means that the *Langseth* turns off all operating airguns.

n. If an observer visually detects a pinniped, the *Langseth* will shut-down the airgun array regardless of the distance of the animal(s) to the sound source. The array will not resume firing until 30 minutes after the last documented pinniped visual sighting.

Resuming Airgun Operations after a Shutdown

o. Following a shutdown, if the observer has visually confirmed that the animal has departed the relevant exclusion zone within a period of less than or equal to 8 minutes after the shutdown, then the *Langseth* may resume airgun operations at full power.

p. Else, if the observer has not seen the animal depart the relevant exclusion zone (with buffer), the *Langseth* shall not resume airgun activity until 15 minutes has passed for species with shorter dive times (*i.e*., small odontocetes) or 30 minutes has passed for species with longer dive durations (*i.e*., mysticetes and large odontocetes, including sperm, pygmy sperm, dwarf sperm, and beaked whales) and pinnipeds. The *Langseth* will follow the ramp-up procedures described in Conditions 6(g) of the IHA.

Survey Operations

q. The *Langseth* may continue marine geophysical surveys into night and low-light hours if the Holder of the Authorization initiates these segment(s) of the survey when the observers can view and effectively monitor the full relevant exclusion zones.

r. This Authorization does not permit the Holder of this Authorization to initiate airgun array operations from a shut-down position at night or during low-light hours (such as in dense fog or heavy rain) when the visual observers cannot view and effectively monitor the full relevant exclusion zones.

s. To the maximum extent practicable, the Holder of this Authorization should schedule seismic operations (*i.e*., firing the airguns) during daylight hours.

t. To the maximum extent practicable, the *Langseth* will conduct the seismic survey (especially when near land) from the coast (inshore) and proceed towards the sea (offshore) in order to avoid trapping marine mammals in shallow water.

Mitigation Airgun

t. 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 airgun would continue to operate. The *Langseth* would not operate the small-volume airgun for longer than three hours in duration during turns.

Special Procedures for Large Whale Concentrations

u. The *Langseth* will avoid concentrations of fin (*Balaenoptera physalus*) and/or sperm whales (*Physeter macrocephalus*) if possible (*i.e*., exposing concentrations of animals to 160 dB re: 1 µPa), and powered-down the array. For purposes of the survey, a concentration or group of whales will consist of six or more individuals visually sighted that do not appear to be traveling (*e.g*., feeding, socializing, etc.). The *Langseth* will follow the procedures described in Conditions 6(k) of the IHA for resuming operations after a power down.

7. Reporting Requirements

This Authorization requires the Holder of this Authorization to:

a. Submit a draft report on all activities and monitoring results to the Chief, Permits and Conservation Division, and Chief, ESA Interagency Cooperation Division, Office of Protected Resources, National Marine Fisheries Service, within 90 days of the completion of the *Langseth's* cruise. This report must contain and summarize the following information:

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

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

   iii. An estimate of the number (by species) of marine mammals with known exposures to the seismic activity (based on visual observation) at received levels greater than or equal to 160 dB re: 1 µPa and/or 180 dB or 190-dB re: 1 µPa for cetaceans and pinnipeds, respectively and a discussion of any specific behaviors those individuals exhibited.

   iv. An estimate of the number (by species) of marine mammals with estimated exposures (based on modeling results) to the seismic activity at received levels greater than or equal to 160 dB re: 1 µPa and/or 180 dB or 190-dB re: 1 µPa with a discussion of the nature of the probable consequences of that exposure on the individuals.

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

b. Submit a final report to the Chief, Permits and Conservation Division, and the Chief, ESA Interagency Cooperation Division, Office of Protected Resources, National Marine Fisheries Service, within 30 days after receiving comments from us on the draft report. If we decide that the draft report needs no comments, we will consider the draft report to be the final report.

8. Reporting Prohibited Take

In the unanticipated event that the specified activity clearly causes the take of a marine mammal in a manner not permitted by the Authorization, such as serious injury or mortality (*e.g.*, ship-strike, gear interaction, and/or entanglement), Lamont-Doherty shall immediately cease the specified activities and immediately report the take to the Chief, Permits and Conservation Division, and the Chief, ESA Interagency Cooperation Division, Office of Protected Resources, NMFS, at 301-427-8400 and/or by email to Jolie.Harrison@noaa.gov and Cathy.Tortorici@noaa.gov.

Lamont-Doherty must also contact the ARION Cetacean Rescue and Rehabilitation Centre, Greece at +030-6945-531850.

The report must include the following information:

- Time, date, and location (latitude/longitude) of the incident;
- Name and type of vessel involved;
- Vessel's speed during and leading up to the incident;
- Description of the incident;
- Status of all sound sources used in the 24 hours preceding the incident;
- Water depth;
- Environmental conditions (*e.g.*, wind speed and direction, Beaufort sea state, cloud cover, and visibility);
- Description of all marine mammal observations in the 24 hours preceding the incident;
- Species identification or description of the animal(s) involved;
- Fate of the animal(s); and
- Photographs or video footage of the animal(s) (if equipment is available).

Lamont-Doherty shall not resume its activities until we are able to review the circumstances of the prohibited take. We shall work with Lamont-Doherty to determine what is necessary to minimize the likelihood of further prohibited take and ensure MMPA compliance. Lamont-Doherty may not resume their activities until notified by us via letter, email, or telephone.

9. Reporting an Injured or Dead Marine Mammal with an Unknown Cause of Death

In the event that Lamont-Doherty discovers an injured or dead marine mammal, and the lead visual 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 we describe in the next section), Lamont-Doherty will immediately report the incident to the Chief, Permits and Conservation Division, and the Chief, ESA Interagency Cooperation Division, Office of Protected Resources, NMFS, at 301-427-8400 and/or by email to Jolie.Harrison@noaa.gov and Cathy.Tortorici@noaa.gov.

Lamont-Doherty must also contact the ARION Cetacean Rescue and Rehabilitation Centre, Greece at +030-6945-531850.

The report must include the same information identified in Condition 8 in the IHA. Activities may continue while we review the circumstances of the incident. We would work with Lamont-Doherty to determine whether modifications in the activities are appropriate.

10. Reporting an Injured or Dead Marine Mammal Unrelated to the Activities

In the event that Lamont-Doherty discovers an injured or dead marine mammal, and the lead visual observer determines that the injury or death is not associated with or related to the authorized activities (*e.g.*, previously wounded animal, carcass with moderate to advanced decomposition, or scavenger damage), Lamont-Doherty would report the incident to the Chief, Permits and Conservation Division, and the Chief, ESA Interagency Cooperation Division, Office of Protected Resources, NMFS, at 301-427-8400 and/or by email to Jolie.Harrison@noaa.gov and Cathy.Tortorici@noaa.gov.

Lamont-Doherty must also contact the ARION Cetacean Rescue and Rehabilitation Centre, Greece at +030-6945-531850.

Lamont-Doherty would provide photographs or video footage (if available) or other documentation of the stranded animal sighting to NMFS.

11. Endangered Species Act Biological Opinion and Incidental Take Statement

Lamont-Doherty must comply with the Terms and Conditions of the Incidental Take Statement corresponding to the Endangered Species Act Biological Opinion issued to the National Science Foundation and NMFS' Office of Protected Resources, Permits and Conservation Division.

A copy of this Authorization and the Incidental Take Statement must be in the possession of all contractors and protected species observers operating under the authority of this Incidental Harassment Authorization.

9 CONSERVATION RECOMMENDATIONS

Section 7(a)(1) of the ESA directs Federal agencies to use their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of the threatened and endangered species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on ESA-listed species or critical habitat, to help implement recovery plans or develop information (50 CFR 402.02).

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 marine mammals as well as endangered or threatened sea turtles and fishes:

1. NMFS encourages the NSF and L-DEO to work with the MOm Hellenic Society for the Study and Protection of the Monk Seal (MOm) or other local experts (e.g., Drs. Alexandros Karamanlidis and Nikolaos Lampadariou) to review and analyze to review and analyze any available data or information to evaluate potential impacts from the seismic survey on Mediterranean monk seals in the Eastern Mediterranean Sea. This could include monitoring known Mediterranean monk seal caves, if possible. Any such information should be included in the report to NMFS.
2. The NSF should promote and fund research examining the potential effects of seismic surveys on listed sea turtle species.
3. The NSF should develop a more robust propagation model that incorporates environmental variables into estimates of how far sound levels reach from airgun sources.

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

10 REINITIATION OF CONSULTATION

This concludes formal consultation for proposed seismic survey to be funded by the NSF and conducted by the L-DEO on board the *R/V Langseth* in the eastern Mediterranean Sea, and NMFS Permits and Conservation Division's issuance of an incidental harassment authorization for the proposed studies pursuant to section 101(a)(5)(D) of the MMPA. As 50 CFR 402.16 states, reinitiation of formal consultation is required where discretionary Federal agency involvement or control over the action has been retained (or is authorized by law) and if: (1) the amount or extent of incidental take is exceeded, (2) new information reveals effects of the agency action that may affect ESA-listed species or critical habitat in a manner or to an extent not considered in this opinion, (3) the agency action is subsequently modified in a manner that causes an effect to the ESA-listed species or critical habitat that was not considered in this opinion, or (4) a new species is ESA-listed or critical habitat designated that may be affected by the action.

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