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: Seismic survey by the National Science Foundation along New Zealand and Issuance of an Incidental Harassment Authorization pursuant to section 101(a)(5)(D) of the Marine Mammal Protection Act

Consultation Conducted By: Endangered Species Act Interagency Cooperation Division, Office of Protected Resources, National Marine Fisheries Service

Approved:

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LIST OF ACRONYMS

ADCP-acoustic Doppler current profiler
CFR-Code of Federal Regulations
CI-confidence interval
CV-coefficient of variation
dB-decibel
DDE-dichlorodiphenyldichloroethylene
DDT-dichlorodiphenyltrichloroethane
ESA-Endangered Species Act
Hz-hertz
hr-hours
IHA-incidental harassment authorization
in-inches
IWC-International Whaling Commission
kHz-kilohertz
km-kilometers
m-meters
min-minutes
MMPA-Marine Mammal Protection Act
NMFS-National Marine Fisheries Service
NOAA-National Oceanic and Atmospheric Administration
NSF-National Science Foundation
Observer-protected species visual observer
PCB-polychlorinated biphenyl
PTS-permanent threshold shift
Revelle-R/V Roger Revelle
RMS-root mean squared
s-seconds
SEL-sound exposure level
SIO-Scripps Institution of Oceanography
t-metric tons
TTS-temporary threshold shift
μPa-microPascals
U.S.-United States
USC-United States Code

1 INTRODUCTION

Section 7(a)(2) of the Endangered Species Act (ESA; 16 U.S.C. 1531 et seq.) requires that each federal agency insure that any action authorized, funded, or carried out by such agency is not likely to jeopardize the continued existence of any endangered or threatened species, or result in the destruction or adverse modification of critical habitat of such species. When the action of a federal agency "may affect" an ESA-listed species or critical habitat designated for it, that agency is required to consult with National Oceanic Atmospheric Administration's (NOAA) National Marine Fisheries Service (NMFS) or the United States (U.S.) Fish and Wildlife Service, depending on the ESA-listed resources that may be affected. For the activities described in this document, the Federal action agencies are the National Science Foundation (NSF) and NMFS's Permits and Conservation Division.

Two federal actions are considered in this biological opinion (Opinion). The first is the NSF's proposal to fund a research vessel, R/V *Roger Revelle* (*Revelle*), which is owned by the U.S. Navy and operated by the Scripps Institution of Oceanography (SIO), to conduct seismic surveys along the east coast of New Zealand (exclusively in New Zealand EEZ waters up to territorial waters) from May to June of 2015, in support of an NSF-funded research project. The second is the NMFS's Permits and Conservation Division's proposal to issue an incidental harassment authorization (IHA) for non-lethal "takes" of marine mammals incidental to the planned seismic surveys, pursuant to section 101 (a)(5)(D) of the Marine Mammal Protection Act (MMPA; 16 U.S.C. 1371 (a)(5)(D)). The consulting agency is the NMFS's Office of Protected Resources, ESA Interagency Cooperation Division.

This document represents NMFS's ESA Interagency Cooperation Division's Opinion on the effects of the two proposed actions on threatened and endangered species, and has been prepared in accordance with section 7 of the ESA. 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.

1.1 Consultation History

On October 27, 2014, the NSF provided basic information regarding a seismic survey along New Zealand and requested.

On October 29, 2014, the NSF, NSF's contractor, NMFS' ESA Interagency Cooperation Division, and NMFS' Permits and Conservation Division discussed the proposed action via conference call. The NMFS' ESA Interagency Cooperation Division suggested data sources to use in estimating marine mammal density and species that may be affected by the proposed action.

On December 9, 2014, the NMFS's ESA Interagency Cooperation Division received a request for formal consultation from the NSF on the effects of its proposed funding of a proposed seismic survey. Information was sufficient to initiate consultation with the NSF on this date.

On December 15, 2014, the NMFS's Permits and Conservation Division received an application from the NSF to incidentally harass marine mammal species during the proposed seismic survey.

On March 12, 2015, the NMFS's ESA Interagency Cooperation Division received a request for formal consultation from the NMFS's Permits and Conservation Division on the effects of its proposed IHA for marine mammal takes during conduct of the same proposed seismic survey. Information was sufficient to initiate consultation with the NSF on this date.

On March 20, 2015, the NMFS's Permits and Conservation Division sent the application for the proposed seismic surveys out to reviewers and published a notice in the *Federal Register* soliciting public comment on their intent to issue an IHA.

2 DESCRIPTION OF THE PROPOSED ACTIONS

Two federal actions are evaluated in this Opinion. The first is the NSF's proposal to fund a research vessel *Revelle*, operated by the SIO, to conduct a seismic survey off the coast of New Zealand from May to June of 2015. 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.[1]

2.1 National Science Foundation Proposed Action

The NSF proposes to fund the use of the U.S. Navy's research vessel *Revelle* to conduct a seismic survey along New Zealand during an approximate 30-day period in mid-May to mid-June, 2015. An array of two operational airguns will be deployed as an energy source. In addition, a multibeam echosounder and sub-bottom profiler will continuously operate from the *Revelle* during the entire cruise. One 600-meter (m)-long hydrophone streamer will also be deployed from the *Revelle*. Several bottom-sampling devices will be lowered to the seafloor and retrieved during the course of the survey.

The proposed study would use the resulting sediment seismic velocities, heat-flow values, and data from sediment cores to develop a process-based understanding of the thermal structure of the Hikurangi subduction zone. Data from sediment cores would detect and estimate the nature and sources of fluid flow through high permeability pathways in the overriding plate and incoming plate, and along the subduction thrust, characterize the hydrocarbon and gas hydrate system to assist with estimates of heat flow from bottom simulating reflectors (BSRs), their role in slope stability, and fluid source, and elucidate the response of microbes involved in carbon cycling to changes in methane flux. This understanding would be expanded by using regional observations of gas hydrate-related BSRs.

2.1.1 Schedule

The NSF proposes to fund the use of the *Revelle*, which is owned by the U.S. Navy and operated by the SIO, for roughly 30 days of operations, including gear setup, deployment, and retrieval and transit from Auckland. The *Revelle* would then transit to the survey area, where all seismic operations will be conducted. The *Revelle* will eventually return to port in Napier, New Zealand. Transit from Auckland to the survey area and the seismic operations are scheduled between 18 May and 18 June, 2015 (roughly 30 days). Some minor deviation from the proposed dates is possible, depending on logistics and weather conditions. Roughly 135 hours of seismic operations are planned, with the balance of time spent in transit or tacking heat flow measurements and ocean floor core samples. Trackline may be reshot and 25% additional trackline has been included in the estimate of total trackline (1,563 km) to account for additional

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[1] Level B harassment is defined under the MMPA as "any act of pursuit, torment, or annoyance which 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."

operations. NMFS's Permits and Conservation Division proposes to issue an IHA that is effective from May 18, 2015 to July 30, 2015.

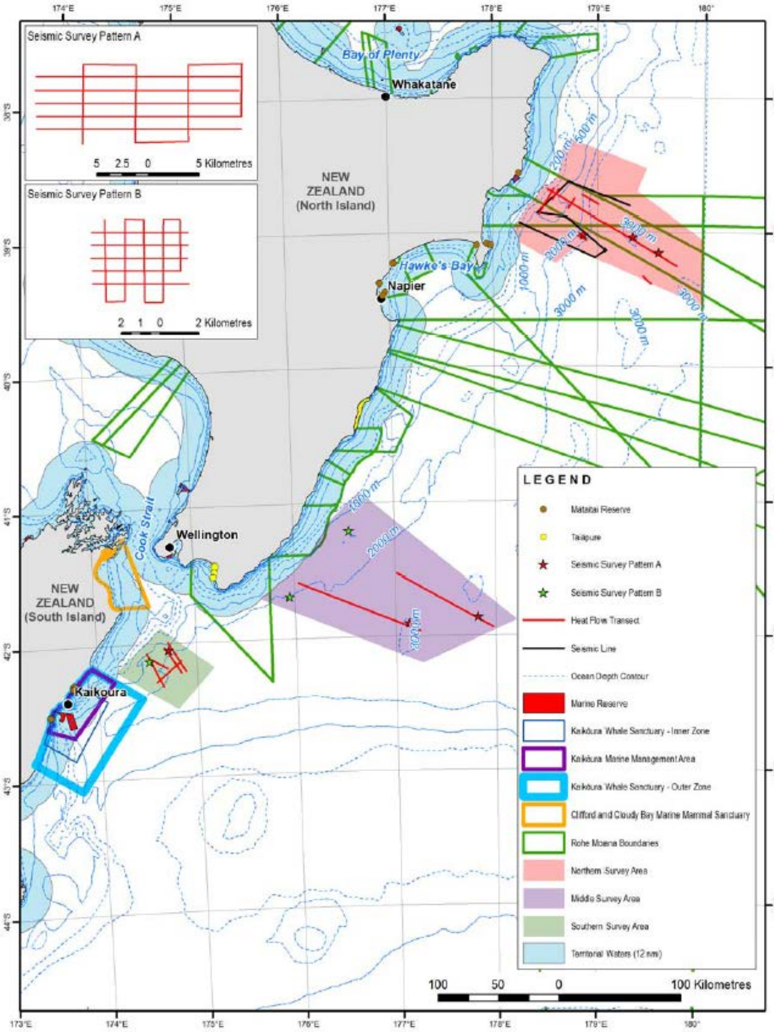

Figure 1. Location of seismic survey along eastern New Zealand.

2.1.2 Source Vessel Specifications

The *Revelle* will tow the airgun array along predetermined lines (**Error! Reference source not found.**). The operating speed during seismic acquisition is typically 9 km/hr (5 knots). When not towing seismic survey gear, the *Revelle* typically cruises at 22-23 km/hr (12-12.5 knots). The *Revelle* also serves as the platform that protected species visual observers (observers) watch for animals.

2.1.3 Airgun Description

The airgun configuration includes two active 45 inches^3 (in) generator-injector (GI) airguns. The airguns will be towed 21 m behind the vessel at a depth of 2 m and fire every 5-10 seconds (s), or every 12.5-25 m travelled. During firing, a brief (approximately 0.1 s) pulse of sound will be emitted. 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.

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

2.1.4 Heat Flow Measurement

The heat-flow probe to be used on the *Revelle* consists of a lance 6 cm in diameter and 3.5 m long, a sensor tube housing thermistors and heater wires, and a 560-kg weight stand. The probe is lowered to the bottom, and a 12-kHz pinger attached to the wire ~50 m above the instrument monitors the distance between the probe and bottom. The probe is driven into the sediment by gravity, and temperatures within the sediment are measured with equally-spaced thermistors. On completion of a measurement, the instrument is hoisted 100–500 m above the sediment, the ship is maneuvered to a new position, and the process is repeated. Heat-flow measurements can generally be made at a rate of 1–2 h per measurement, about 15 min for the actual measurement and 45–90 min to reposition the ship and probe.

2.1.5 Piston Core and Gravity Core Measurements

The piston corer to be used on the *Revelle* consists of a piston core with a 10 cm diameter steel barrel up to 18 m long with a 2,300 kg weight and a trigger core with a 10 cm diameter PVC plastic barrel 3 m long with a 230 kg weight, which are lowered concurrently into the ocean floor with 1.4-cm diameter steel cables.

The gravity corer consists of a 6-m long core pipe that takes a core sample about 10 cm in diameter, a head weight about 45 cm in diameter, and a stabilizing fin. It is lowered to the ocean floor with 1.4 cm diameter steel cable at 100 m/min speed.

2.1.6 Multibeam Echosounder and Sub-bottom Profiler

Along with airgun operations, additional acoustical data acquisition systems will operate during the surveys from the *Revelle*. The multibeam echosounder as well as sub-bottom profiler systems will map the ocean floor during the cruise. These sound sources will operate from the *Revelle* simultaneously with the airgun array, as well as when the airguns are shutdown.

The sub-bottom profiler (Knudsen 3260) is a hull-mounted sonar system that operates at 3.5 with a single 27° bottom-directed beam. The nominal power output is 10 kilowatts, but the actual maximum radiated power is 3 kilowatts or 222 dB re: 1 μPa·m. The ping duration is up to 64 milliseconds, and the ping interval is 1 s. A common mode of operation is to broadcast five pings at 1 s intervals.

The multibeam echosounder (Kongsberg EM 122) is also a hull-mounted system operating at 12 kHz. The beam width is 1 or 2° fore–aft and 150° perpendicular to the ship's line of travel. The maximum source level is 242 dB re: 1 $\mu Pa \cdot m_{rms}$. Each "ping" consists of four or eight successive fan-shaped transmissions, each 2-15 milliseconds in duration and each ensonifying a sector that extends 1° fore–aft. Four or eight successive transmissions span an overall cross-track angular extent of about 150°.

2.1.7 Proposed Exclusion Zones

The NSF will implement exclusion zones around the *Revelle* to minimize potential adverse effects of airgun sound on MMPA and ESA-listed species. These zones are areas where seismic airguns are shut-down to reduce exposure of marine mammals to sound levels have the potential to result in injury. These exclusion zones are based on modeled sound levels at various distances from the *Revelle,* described below. Normally, the exclusion zone is based on isopleth modeling from the acoustic source to the 180 dB re: 1 μPa_{rms} isopleth.

Predicted Sound Levels vs. Distance and Depth

The Lamont-Doherty Earth Observatory has predicted received sound levels, in relation to distance and direction from two 45-in^3 GI airguns in deep water (Figure 2). Empirical data concerning 190, 180, and 160 dB re: 1 μPa_{rms} distances were acquired during the acoustic calibration study of the *R/V Ewing*'s airgun array in a variety of configurations in 2003 in the northern Gulf of Mexico (Tolstoy et al. 2004) and in 2007-2009 aboard the *R/V Langseth* (Diebold et al. 2010; Tolstoy et al. 2009). As a two-airgun array at the same tow and water depths were not measured, the estimates provided here were extrapolated from other results, using conservative assumptions. Results of the propagation measurements (Tolstoy et al. 2009) showed that radii around the airguns for various received levels varied with water depth. However, the depth of the array was different in the Gulf of Mexico calibration study (6 m) from in the proposed survey (2-3 m). Because propagation varies with array depth, correction factors have been applied to the distances reported by Tolstoy et al. (2009).

Table 1 shows the distances at which four rms (root mean squared) sound levels are expected to be received from the airgun array. The 180 dB re: 1 μPa_{rms} distance is the safety criteria as specified by NMFS (1995) for cetaceans, as required by the NMFS during most other recent Lamont-Doherty Earth Observatory seismic projects (Holst and Beland 2008; Holst and Smultea 2008b; Holst et al. 2005a; Holt 2008; Smultea et al. 2004).

Table 1. Predicted distances to which sound levels ≥190, 180, 166 and 160 dB re: 1 μPa$_{rms}$ could be received from the two-airgun, 90 in^3 array towed at 2 m.

Water depth	Predicted rms radii (m)			
	190 dB	180 dB	166 dB	160 dB
>1,000 m	10	100	200	400
100-1,000 m	15	100	300	600

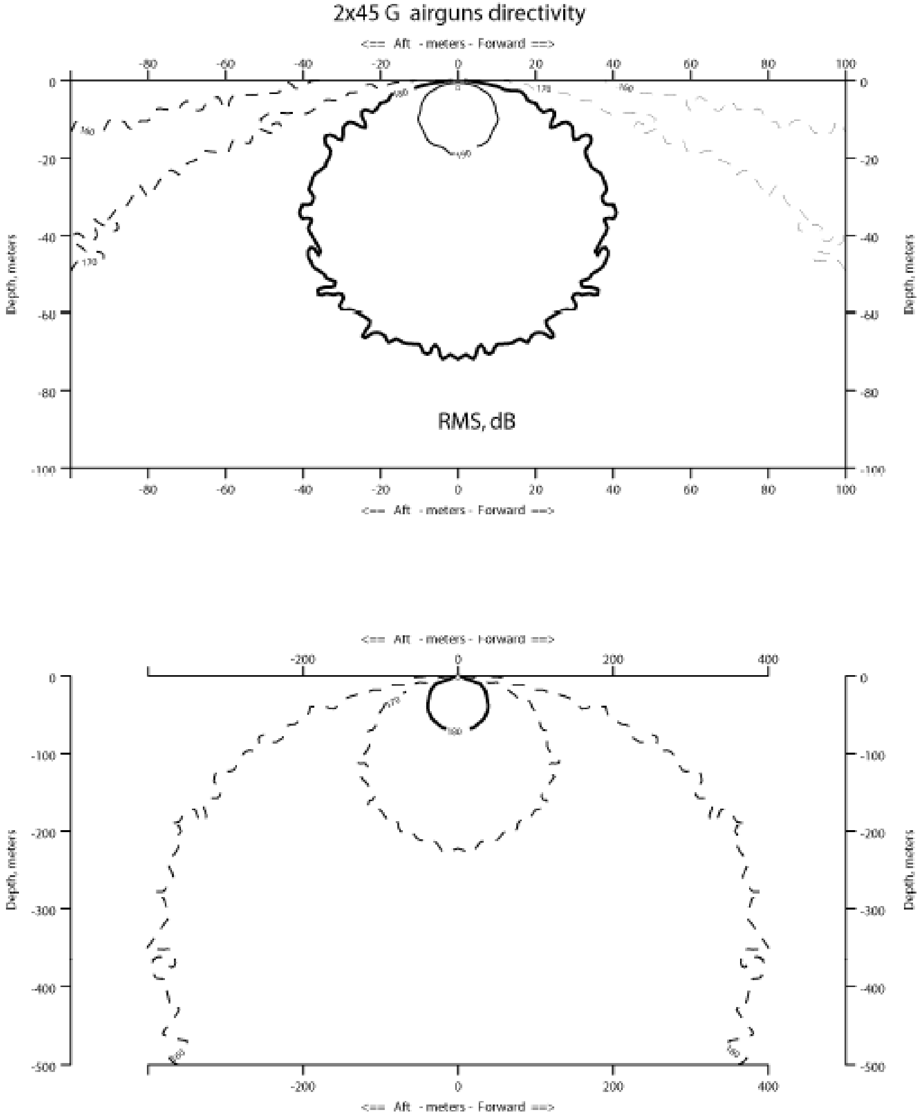

Figure 2. Modeled received sound levels (SELs) from two 45-in^3 G airguns operating in deep water at a 2-m tow depth. Received rms levels (SPLs) are likely ~10 dB higher.

2.2 NMFS Permits and Conservation Division's Incidental Harassment Authorization

The NMFS's 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 by SIO. The IHA will be valid from May 18, 2015 to July 30, 2015, and will authorize the incidental harassment of the following endangered species: blue whales (*Balaenoptera musculus*), fin whales (*Balaenoptera physalus*), humpback whales (*Megaptera novaeangliae*), sei whales (*Balaenoptera borealis*), southern right whales (*Eubalaena australis*), sperm whales (*Physeter macrocephalus*), and other non-listed marine mammals. The IHA will not authorize incidental takes in the territorial seas (within 12 nautical miles) of New Zealand. The IHA would state the following:

SIO must adhere to the following conditions for the IHA to remain valid:

1. Establish an exclusion zone[2] corresponding to the anticipated 180 dB re: 1 µParms isopleth for airgun array operations as well as a 160 dB re: 1 µParms buffer zone.
2. Use one NMFS-approved, vessel-based observer to watch for, and monitor, marine mammal species near the seismic source vessel during daytime airgun operations (nautical twilight-dawn to nautical twilight-dusk), and while the seismic array and streamers are being deployed and retrieved. When practical, vessel crew will also assist in detecting marine mammals. Protected species visual observer (observers) will have access to reticle binoculars (7 X 50 Fujinon), big-eye binoculars (25 x 150), optical range finders, and night-vision devices. Observer’s shifts will last no longer than four hours at a time. When feasible, observers will also watch during daytime periods when the seismic system is not operating, for comparisons of animal abundance and behavior.
3. Record the following information when a marine mammal is sighted:
    a. 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.
    b. Time, location, heading, speed, activity of the vessel (including number of airguns operating and whether in state of ramp-up or power-down), Beaufort sea state and wind force, visibility, cloud cover, and sun glare.
    c. The data listed under b. would also be recorded at the start and end of each observation watch and during a watch whenever there is a change in one or more of the variables.
4. Use two observers to watch the entire exclusion zone, for at least 30 minutes (min) before starting the airgun (day or night). If observers find a marine mammal within the exclusion zone, SIO must delay the seismic survey until the marine mammal has left the area. If the observer sees a marine mammal that surfaces, then dives below the surface, the observer shall wait 30 min. During that time, if the observer sees no marine mammals they should assume that the animal has moved beyond the exclusion zone. If, for any reason, the entire radius cannot be seen for the entire 30 min (e.g., rough seas, fog, darkness), or if marine mammals are near, approaching or in the exclusion zone, the airguns may not be started up.

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[2] The “exclusion zone” refers to a region around the seismic airgun source where mitigation would be undertaken to avoid or minimize the impacts of the airguns if marine mammals are observed within it.

If one airgun is already running at a source level of at least 180 dB re: 1 µPa$_{rms}$, SIO may start the second gun without observing the entire exclusion zone for 30 min prior, provided no marine mammals are known to be near the safety radius.

5. Apply a "ramp-up" procedure when starting up at the beginning of seismic operations or any time after the entire array has been shut-down for more than 15 min. This procedure means starting the smallest gun first and adding the second airgun in a sequence such that the source level of the array will increase in steps not exceeding approximately 6 dB per 5-min period. During ramp-up, the two observers will monitor the 180 dB re: 1 µParms exclusion zone, and if marine mammals are sighted, a course/speed alteration or shut-down will occur as though the full array were operational.
6. Alter speed or course during seismic operations if a marine mammal, based on its position and relative motion, appears likely to enter the exclusion zone. If speed or course alteration is not safe or practical, or, if after alteration, the marine mammal still appears likely to enter the exclusion zone, further mitigation measures, such as shut-down, will be taken.
7. Shutdown the airguns whether marine mammal are detected within, approaching, or entering the exclusion zone. Airgun activity will not resume until the marine mammal has cleared the exclusion zone, which means it was seen leaving the exclusion zone, or has not been seen within the exclusion zone for 15 min (small odontocetes) or 30 min (mysticetes and large odontocetes).
8. Marine seismic operations may continue into night and low-light hours if segment(s) of the survey is started when the entire exclusion zone is visible and can be effectively monitored. Do not start 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 entire exclusion zone cannot be effectively monitored by the observer(s) on duty. To the maximum extent practicable, seismic airgun operations should be scheduled during daylight hours.
9. If the specified activity clearly causes any unanticipated cases of marine mammal injury or mortality resulting from these activities (e.g., ship-strike, gear interaction, and/or entanglement), SIO will cease operating seismic airguns and report the incident to NMFS's Office of Protected Resources at 301-427-8401 and/or by email to *Jolie.Harrison@noaa.gov* and *Howard.Goldstein@noaa.gov,* immediately.

   The report must include:

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

Airgun operation will then be postponed until NMFS is able to review the circumstances and then work with SIO to determine whether modifications in the activities are appropriate and necessary.

10. If SIO discovers an injured or dead marine mammal, and the lead observer determines both the cause of the injury or death is unknown and the death is relatively recent (i.e., in less than a moderate state of decomposition as described in the next paragraph), NSF will immediately report the incident to the Chief of the Permits and Conservation Division, Office of Protected Resources, NMFS, at 301-427-8401, and/or by email to *Jolie.Harrison@noaa.gov* and *Howard.Goldtein@noaa.gov*. The report must include the same information as described above. Activities may continue while NMFS reviews the circumstances of the incident. NMFS will work with SIO to determine whether modifications in the activities are appropriate.
11. If SIO 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 activities (e.g., previously wounded animal, carcass with moderate to advanced decomposition, or scavenger damage), SIO shall report the incident to the Chief of the Permits and Conservation Division, at the address above, within 24 hr of the discovery. SIO shall provide photographs or video footage (if available) or other documentation of the animal sighting to NMFS. Activities may continue while NMFS reviews the circumstances of the incident.
12. SIO is required to comply with the Terms and Conditions of this Opinion’s Incidental Take Statement issued to both the SIO and the NMFS’s Office of Protected Resources.

In addition, the proposed incidental harassment authorization requires SIO to obey the following reporting requirements:

1. SIO is required to submit a report on all activities and monitoring results to the Office of Protected Resources, NMFS, within 90 days after the completion of the *Revelle*’s cruise.
    a. 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.
    b. 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.
    c. An estimate of the number (by species) of marine mammals that:
        i. Are known to have been exposed to the seismic activity (visual observation) at received levels greater than or equal to 160 dB re: 1 microPa (rms) and/or 180 dB re: 1 microPa (rms) for cetaceans during seismic airgun operations, with a discussion of any specific behaviors those individuals exhibited.
        ii. May have been exposed (modeling results) to the seismic activity at received levels greater than or equal to 160 dB re: 1 microPa (rms) and/or 180 dB re: 1 microPa (rms) with a discussion of the nature of the probable consequences of that exposure on the individuals that have been exposed.
    d. A description of the implementation and effectiveness of the:

i. Terms and conditions of the Opinion’s Incidental Take Statement.

ii. Mitigation measures of the IHA: For the Opinion, the report will confirm the implementation of each term and condition and describe the effectiveness, as well as any conservation measures, for minimizing the adverse effects of the action on listed whales.

3 APPROACH TO THE ASSESSMENT

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

The final steps of our analyses – establishing the risks those responses pose to ESA-listed resources – are different for ESA-listed species and designated critical habitat (these represent our *risk analyses*). Our jeopardy determinations must be based on an action's effects on the continued existence of threatened or endangered species as those "species" have been listed, which can include true biological species, subspecies, or distinct population segments of vertebrate species. The continued existence of these "species" depends on the fate of the populations that comprise them. Similarly, the continued existence of populations are determined by the fate of the individuals that comprise them – populations grow or decline as the individuals that comprise the population live, die, grow, mature, migrate, and reproduce (or fail to do so).

Our risk analyses reflect these relationships between ESA-listed species, the populations that comprise that species, and the individuals that comprise those populations. Our risk analyses begin by identifying the probable risks actions pose to ESA-listed individuals that are likely to be exposed to an action's effects. Our analyses then integrate those individual risks to identify consequences to the populations those individuals represent. Our analyses conclude by determining the consequences of those population-level risks to the species those populations comprise.

We measure risks to ESA-listed individuals using the individuals' "fitness," or the individual's growth, survival, annual reproductive success, and lifetime reproductive success. In particular, we examine the scientific and commercial data available to determine if an individual's probable lethal, sub-lethal, or behavioral responses to an action's effect on the environment (that we identify during our response analyses) are likely to have consequences for the individual's fitness.

When an individual is expected to experience reductions in fitness in response to an action's effects, those fitness reductions may reduce the abundance, reproduction, or growth rates (or increase the variance in these measures) of the populations those individuals represent (see Stearns 1992). Reductions in at least one of these variables (or one of the variables we derive from them) is a *necessary* condition for reductions in a population's viability, which is itself a *necessary* condition for reductions in a species' viability. As a result, when ESA-listed plants or animals exposed to an action's effects are *not* expected to experience reductions in fitness, we would not expect the action to have adverse consequences on the viability of the populations those individuals represent or the species those populations comprise (e.g., Anderson 2000; Brandon 1978; Mills and Beatty 1979; Stearns 1992). If we conclude that ESA-listed plants or animals are *not* likely to experience reductions in their fitness, we would conclude our assessment.

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

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

To conduct these analyses, we rely on all of the best scientific and commercial evidence available to us. This evidence consists of the environmental assessment submitted by the NSF, monitoring reports submitted by past and present seismic survey operators, reports from NMFS Science Centers; reports prepared by natural resource agencies in states and other countries, reports from non-governmental organizations involved in marine conservation issues, the information provided by NMFS's Permits and Conservation Division when it initiates formal consultation, the general scientific literature, and our expert opinion.

During the consultation, we conducted electronic searches of the general scientific literature using search engines, including Agricola, Ingenta Connect, Aquatic Sciences and Fisheries Abstracts, JSTOR, Conference Papers Index, First Search (Article First, ECO, WorldCat), Web of Science, Oceanic Abstracts, Google Scholar, and Science Direct. We also referred to an internal electronic library that represents a major repository on the biology of ESA-listed species under the NMFS's jurisdiction.

We supplemented these searches with electronic searches of doctoral dissertations and master's theses. These searches specifically tried to identify data or other information that supports a particular conclusion (for example, a study that suggests whales will exhibit a particular response to acoustic exposure or close vessel approach) as well as data that do not support that conclusion. When data are equivocal or when faced with substantial uncertainty, our decisions are designed to avoid the risks of incorrectly concluding that an action would not have an adverse effect on ESA-listed species when, in fact, such adverse effects are likely (i.e., Type II error).

3.1 Assessment Approach Applied to this Consultation

In this particular assessment, we identified the potential stressors associated with the action and determined which were probable based on 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 airguns 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) (Figure 3). However, it should not be assumed that anthropogenic stressors lead to fitness consequences at the individual or population levels (New et al. 2013).

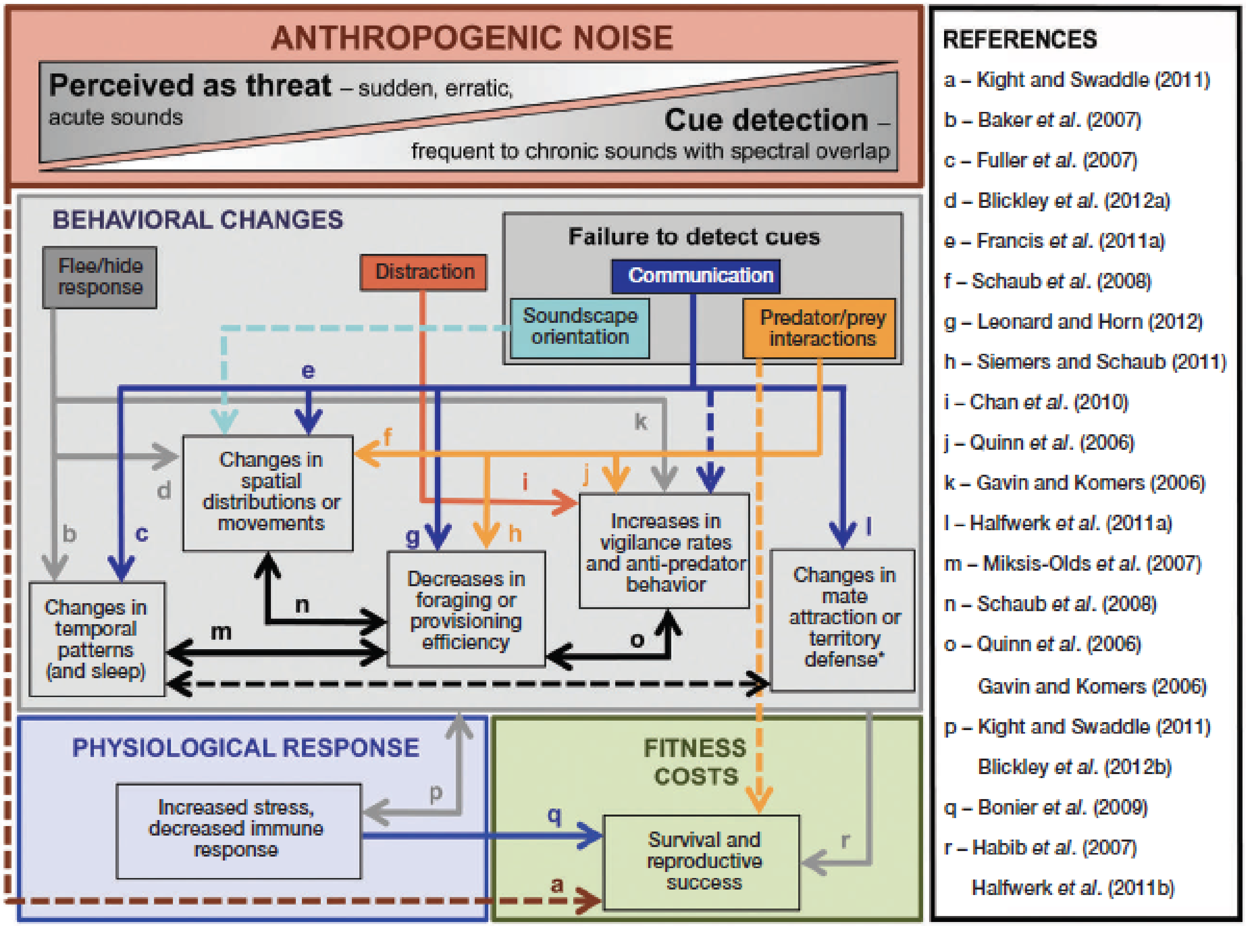

Figure 3. Conceptual framework of how anthropogenic noise affects individuals and how those impacts may lead to fitness consequences.[3]

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. 2007a) better than high frequencies (Southall et al. 2007a), as the former is primarily the range in which they vocalize. Humpback whales frequently vocalize with mid-frequency sound (Southall et al. 2007a) and are likely to hear at these frequencies as well. Because of this, we can partition baleen whales into two groups: those that are specialists at hearing low frequencies (e.g., fin 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

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[3] Figure taken from Francis et al. Francis, C. D., and J. R. Barber. 2013. A framework for understanding noise impacts on wildlife: An urgent conservation priority. Frontiers in Ecology and the Environment 11(6):305-313.

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).

4 ACTION AREA

The seismic survey will occur along the east coast of both North and South Islands of New Zealand (Figure 4), in the Exclusive Economic Zone waters of New Zealand. Some sound from airguns may enter into the territorial waters of New Zealand. Although we do assess the effects of this, we do not authorize take in the territory of foreign nations such as this.

The region in which the seismic survey will occur is between 38.5° and 42.5° S and 174° and 180° E. The region encompasses water depths from 200-3,000 m along roughly 1,563 km of trackline. The action area also includes transit from Auckland and to Napier, New Zealand as well as the region that sound from the seismic survey vessels and their sound sources decrease to ambient background levels.

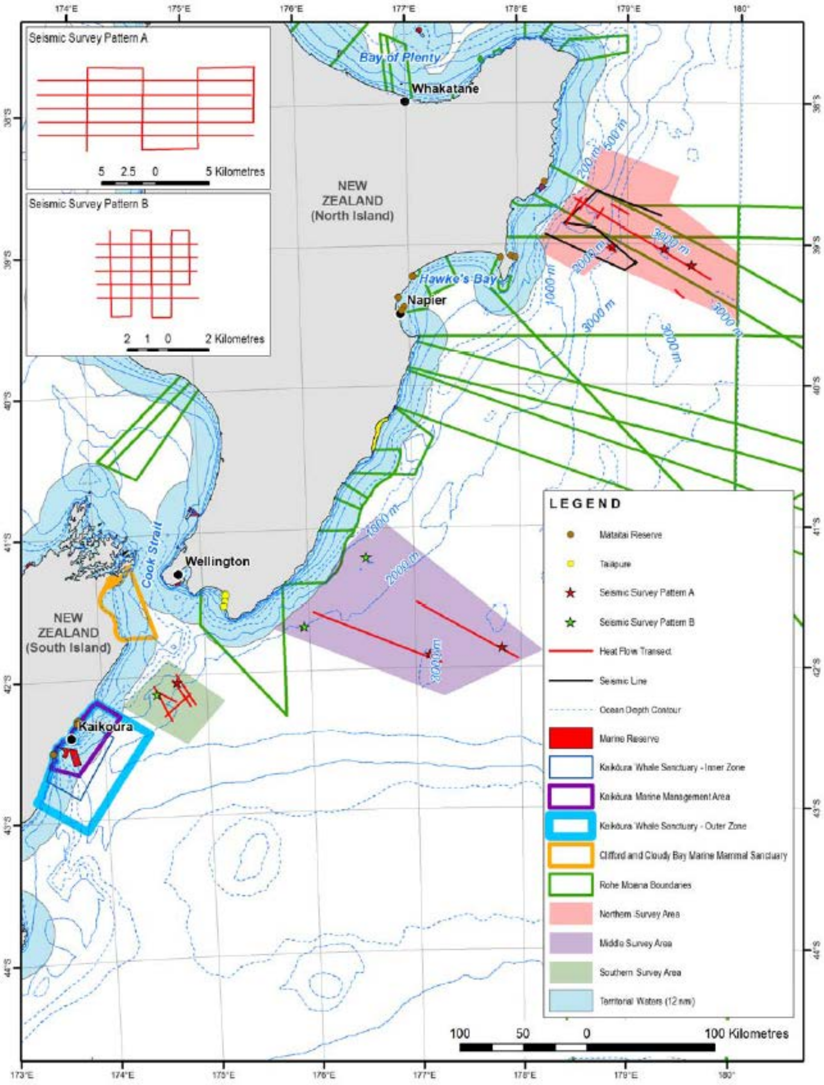

Figure 4. Proposed area for the marine seismic survey along New Zealand.

5 STATUS OF LISTED RESOURCES

The actions considered in this Opinion may affect ESA-listed species in Table 2.

Table 2. Species listed under the Endangered Species Act (ESA) in the action area that may be affected by the proposed actions.

Common Name	*Scientific Name*	*Status*	*Listing*	*Critical Habitat*	*Recovery Plan*
Cetaceans					
Blue whale	*Balaenoptera musculus*	E	35 FR 18319 12/2/1970	None designated	7/1998*
Fin whale	*Balaenoptera physalus*	E	35 FR 18319 12/2/1970	None designated	75 FR 47538 8/6/2010
Humpback whale	*Megaptera novaeangliae*	E	35 FR 18319 12/2/1970	None designated	11/1991*
Sei whale	*Balaenoptera borealis*	E	35 FR 18319 12/2/1970	None designated	12/2011*
Southern right whale	*Eubalaena australis*	E	35 FR 8491 06/02/1970	None designated	Not available
Sperm whale	*Physeter macrocephalus*	E	35 FR 18319 12/2/1970	None designated	75 FR 81584 12/28/2010
Marine Turtles					
Green sea turtle	*Chelonia mydas*	T	43 FR 82300 7/28/1978	63 FR 46693 9/2/1998	63 FR 28359 5/22/1998
Hawksbill sea turtle	*Eretmochelys imbricata*	E	35 FR 8491 06/02/1970	None designated	63 FR 28359 05/22/1998
Leatherback sea turtle	*Dermochelys coriacea*	E	35 FR 8491 5/2/1970	44 FR 17710 3/23/1979	Not available†
Loggerhead sea turtle – South Pacific DPS	*Caretta caretta*	E	76 FR 58868 9/22/2011	None designated	10/29/1991*
Olive ridley sea turtle	*Lepidochelys olivacea*	T	43 FR 32800 07/28/1978	None designated	63 FR 28359 05/22/1998

*FR Notice information not available.

5.1 Blue whale

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

intermedia and *B. m. brevicauda* does apparently occur (Attard et al. 2012). Both pygmy and true blue whales occur in New Zealand waters (Branch et al. 2007a).

Population structure. Little is known about population and stock structure[4] of blue whales. Studies suggest a wide range of alternative population and stock scenarios based on movement, feeding, and acoustic data. Some suggest that as many as 10 global populations, while others suggest that the species is composed of a single panmictic population (Gambell 1979; Gilpatrick and Perryman. 2009; Reeves et al. 1998). For management purposes, the International Whaling Commission (IWC) considers all Pacific blue whales to be a single stock, whereas under the MMPA, the NMFS recognizes four stocks of blue whales: western North Pacific Ocean, eastern North Pacific Ocean, Northern Indian Ocean, and Southern Hemisphere.

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

Southern Hemisphere. During the austral summer, blue whales range from the edge of the Antarctic pack ice (40°-78° S) and during the austral winter, north to Ecuador, Brazil, South Africa, Australia, and New Zealand (Shirihai 2002). Occurrence in Antarctic waters appears to be highest from February to May as well as in November (Gedamke and Robinson 2010; Sirovic et al. 2009). Gedamke and Robinson (2010) found blue whales to be particularly numerous and/or vocal north of Prydz Bay, Antarctica, based on sonobuoy deployments. Pygmy blue whales were also frequently heard in Antarctic waters, further south than previously documented (Gedamke and Robinson 2010). Other than a single vocal record in Atlantic waters, off Angola, pygmy blue whales have been exclusively documented in the Indian or western Pacific Ocean (Cerchio et al. 2010; Mccauley and Jenner 2010). A strong male bias may exist in true blue whales (Double et al. 2013).

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

Blue whales are occasionally sighted in pelagic waters off the western coast of Costa Rica and Nicaragua, near the Galápagos Islands, and along the coasts of Ecuador and northern Peru (Aguayo 1974; Clarke 1980b; Donovan 1984; LGL Ltd. 2007; Mate et al. 1999; Palacios 1999; Reilly and Thayer 1990). Individuals here may represent two populations; the true and pygmy blue whales of the Southern Hemisphere (Gilpatrick and Perryman. 2009), although, recent analyses of vocalizations and photos have linked blue whales found in the Costa Rica Dome to the North Pacific population (Chandler and Calambokidis 2004). Other individuals are linked via song comparisons to blue whales foraging along southern Chile (Buchan et al. 2014). Blue whales appear to be present off New Zealand year round (Miller et al. 2014), where a foraging ground may be present (Torres et al. 2013e).

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

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

Habitat. Blue whales occur in New Zealand waters throughout the year, primarily in South Taranaki Bight, along the east coast of Northland, East Cape, and Bay of Plenty and mainly during the austral spring and summer (including in the action area) (Berkenbush et al. 2013; Clement 2010; Torres 2013). Additional vocal detections have been made from March through December (McDonald 2006; Miller et al. 2013). The South Taranaki Bight appears to be a foraging area (Torres 2013). Numerous stranded true and pygmy blue whales have been documented in New Zealand (Berkenbush et al. 2013; Torres 2013).

Movement. Satellite tagging indicates that, for blue whales tagged off Southern California, movement is more linear and faster (3.7 km/h) while traveling versus while foraging (1.7 km/h)(Bailey et al. 2009). Residency times in what are likely prey patches average 21 days and constituted 29% of an individual’s time overall, although apparently foraging could occur at any time of year for tagged individuals (Bailey et al. 2009). Broad scale movements also varied greatly, likely in response to oceanographic conditions influencing prey abundance and distribution (Bailey et al. 2009). Blue whales along Southern California were found to be traveling 85% of the time and milling 11% (Bacon et al. 2011). In the Southern Ocean, blue whales travel long distances, but frequently concentrate in small areas, presumably searching for or feeding on prey, particularly along the ice edge (Andrews-Goff et al. 2013). Blue whales are highly mobile, and their migratory patterns are not well known (Perry et al. 1999; Reeves et al.

2004). In fall, blue whales migrate toward the warmer waters of the subtropics to reduce energy costs, avoid ice entrapment, and reproduce (NMFS 1998a). In the eastern Central Atlantic, blue whales appear to migrate from areas along Greenland and Iceland to the Azores over and east of the Mid-Atlantic Ridge, apparently engaging in some random movement along the way (Anil et al. 2013).

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

Vocalization and hearing. Blue whales produce prolonged low-frequency vocalizations that include moans in the range from 12.5-400 Hertz (Hz), with dominant frequencies from 16-25 Hz, and songs that span frequencies from 16-60 Hz that last up to 36 sec repeated every 1 to 2 min (see Cummings and Thompson 1971; Cummings and Thompson 1977; Edds-Walton 1997a; Edds 1982; McDonald et al. 1995a; Thompson and Friedl 1982). Non-song vocalization are also low-frequency in nature (generally below 200 Hz, but one of six types up to 750 Hz) between 0.9 and 4.4 s long (Redalde-Salas et al. 2014). Berchok et al. (2006) examined vocalizations of St. Lawrence blue whales and found mean peak frequencies ranging from 17.0-78.7 Hz. Reported source levels are 180-188 dB re: 1µPa, but may reach 195 dB re: 1µPa (Aburto et al. 1997; Clark and Ellison 2004; Ketten 1998; McDonald et al. 2001). Samaran et al. (2010) estimated Antarctic blue whale calls in the Indian Ocean at 179 ± 5 dB re: 1 μPa_{rms} at 1 m in the 17-30 Hz range and pygmy blue whale calls at 175± 1 dB re: 1 μPa_{rms} at 1 m in the 17-50 Hz range. Source levels around Iceland have been 158-169 dB re: 1 μPa_{rms} (Rasmussen et al. 2013). Direct studies of blue whale hearing have not been conducted, but it is assumed that blue whales can hear the same frequencies that they produce (low-frequency) and are likely most sensitive to this frequency range (Ketten 1997; Richardson et al. 1995c).

Vocalizations attributed to blue whales have been recorded in presumed foraging areas, along migration routes, and during the presumed breeding season (Beamish and Mitchell 1971; Cummings et al. 1972a; Cummings and Thompson 1971; Cummings and Thompson 1977; Cummings and Thompson 1994; Rivers 1997; Thompson et al. 1996). Blue whale calls appear to vary between western and eastern North Pacific regions, suggesting possible structuring in populations (Rivers 1997; Stafford et al. 2001).

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

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

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

Southern Hemisphere. Estimates of 4-5% for an average rate of population growth have been proposed (Yochem and Leatherwood 1985). However, a recent estimate of population growth for Antarctic blue whales throughout the region was 7.3% (Branch et al. 2007b). Punt (2010) estimated the rate of increase for blue whales in the Southern Hemisphere to be 8.2% annually (3.37 standard error) between 1978 and 2004. Antarctic blue whales remain severely depleted with the 1996 estimate only 0.7% of pre-whaling levels (IWC 2005). Blue whales along Chile have been estimated to number between 7 and 9% of historical abundance (Williams et al. 2011). Genetic diversity remains reasonable to high here considering the extreme bottleneck that the population experienced (Sremba et al. 2012; Torres-Florez et al. 2014).

Table 3. Summary of past and present blue whale abundance

Region	Population, stock, or study area	Pre-exploitation estimate	95% CI	Current estimate	95% CI	Source
Global	~~	200,000	~~	11,200-13,000	~~	(DOC 1983; Maser et al. 1981)
	~~	~~	~~	5,000-12,000	~~	(COSEWIC 2002)
Southern Hemisphere	Basinwide	150,000-210,000		5,000-6,000	~~	(Gambell 1976; Yochem and Leatherwood 1985)
	~~	300,000	~~	~~	~~	(COSEWIC 2002)
	~~	~~	~~	400-1,400	400-1,400	IWC, for years 1980-2000
	~~	~~	~~	1,700	860-2,900	(IWC 2005c), point estimate for 1996
	Within IWC survey areas	~~	~~	1,255	~~	(IWC 1996)
	~~	10,000	~~	5,000	~~	(Gambell 1976)

Region	Population, stock, or study area	Pre-exploitation estimate	95% CI	Current estimate	95% CI	Source
	~~	13,000	~~	6,500	~~	(Zemsky and Sazhinov 1982)
				2,300 true		IWC (2014)
				1,500 pygmy		Boyd (2002)
	South of 60° S			1,700		(Branch et al. 2007b)

**Note: CIs not provided by the authors were calculated from CVs where available, using the computation from Gotelli and Ellison (2004).*

After the explosive harpoon was developed in the late nineteenth century, blue whales were the mainstay of whaling in the region (Shirihai 2002). During the early 1900s, the species became a principal target of the whaling industry throughout the world, with the majority killed in the Southern Hemisphere. Approximately 330,000–360,000 blue whales were harvested from 1904 to 1967 in the Antarctic alone, reducing their abundance to <3% of their original numbers (Perry et al. 1999; Reeves et al. 2003). Blue whales were protected in portions of the Southern Hemisphere beginning in 1939, and received full protection in the Antarctic in 1966. During two separate surveys, 24 (Ensor et al. 2003) and 30 (Smith Jr. et al. 2012) individuals were observed in the Ross Sea.

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

Anthropogenic threats. Blue whales have faced threats from several historical and current sources. Blue whale populations were severely depleted originally due to historical whaling activity. Ship strike is an issue for blue whales of Sri Lanka engaged in foraging in shipping lanes, with several individuals stranding or being found with evidence of being struck (De Vos et al. 2013; Ilangakoon 2012). Additional mortality from ship strikes probably goes unreported because the whales do not strand or, if they do, they do not always have obvious signs of trauma. Studies have shown that blue whales respond to approaching ships in a variety of ways, depending on the behavior of the animals at the time of approach, and speed and direction of the approaching vessel. While feeding, blue whales react less rapidly and with less obvious avoidance behavior than whales that are not feeding (Sears 1983).

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

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

5.2 Fin whale

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

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

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

Age distribution. Aguilar and Lockyer (1987) suggested annual natural mortality rates in northeast Atlantic fin whales may range from 0.04 to 0.06. This is supported by an estimated annual survival rate of 0.955 for Gulf of St. Lawrence fin whales (Ramp et al. 2014). Fin whales live 70-80 years (Kjeld et al. 2006).

Habitat. Spring and summer of the primary time that fin whales occur in New Zealand waters. This includes sighting records in the action area and surrounding vicinity (Barker et al. 2009; Berkenbush et al. 2013; Clement 2010) as well as vocal recordings from the surrounding region(McDonald 2006). More than a dozen fin whales have been recorded stranded on New Zealand shores (Brabyn 1991).

Reproduction. Fin whales reach sexual maturity between 5-15 years of age (COSEWIC 2005; Gambell 1985a; Lockyer 1972). Mating and calving occurs primarily from October-January, gestation lasts ~11 months, and nursing occurs for 6-11 months (Boyd et al. 1999; Hain et al. 1992). The average calving interval in the North Atlantic is estimated at about 2-3 years (Agler et al. 1993; Christensen et al. 1992). The location of winter breeding grounds is uncertain but mating is assumed to occur in pelagic mid-latitude waters (Perry et al. 1999). This was recently contradicted by acoustic surveys in the Davis Strait and off Greenland, where singing by fin whales peaked in November through December; the authors suggested that mating may occur before southbound migration (Simon et al. 2010). Although seasonal migration occurs between presumed foraging and breeding locations, fin whales have been heard 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). Recent IWC scientific whaling data suggest that, compared to commercial whaling periods, pregnancy rates have decreased, age at sexual maturity has increased, size growth is slowing, and males now compose a slightly higher proportion of the population than female (Gunnlaugsson et al. 2013).

Behavior. 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. 2009; Nasu 1974).

In waters off the U.S. Atlantic Coast, individuals or duos represented about 75% of sightings (Hain et al. 1992). Individuals or groups of fewer than five individuals represent about 90% of observations. Barlow (2003) reported mean group sizes of 1.1–4.0 during surveys off California, Oregon, and Washington.

Vocalization and hearing. Fin whales produce a variety of low-frequency sounds in the 10-200 Hz range (Edds 1988; Thompson et al. 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 s 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 1997b; 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 blue whales can hear the same frequencies that they produce (low) and are likely most sensitive to this frequency range (Ketten 1997; Richardson et al. 1995c).

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

Table 4. Summary of past and present fin whale abundance

Region	Population, stock, or study area	Pre-exploitation estimate	95% CI	Recent estimate	95% CI	Source
Global	~~	>464,000	~~	119,000	~~	(Braham 1991)
Southern Hemisphere	Basinwide	400,000	~~	85,200	~~	(Braham 1991; IWC 1979)
	South of 60°S	~~	~~	1,735	514-2,956	(IWC 1996)
	South of 30°S	~~	~~	15,178	~~	(IWC 1996)
				15,000		Boyd (2002)
	Scotia Sea and Antarctic Peninsula	~~	~~	4,672	792-8,552	(Hedley et al. 2001; Reilly et al. 2004b)

**Note: CIs not provided by the authors were calculated from CVs where available, using the computation from Gotelli and Ellison (2004).*

Southern Hemisphere. The Southern Hemisphere population was one of the most heavily exploited whale populations under commercial whaling. From 1904 to 1975, over 700,000 fin whales were killed in Antarctic whaling operations (IWC 1990). Harvests increased substantially by the introduction of factory whaling ships in 1925, with an average of 25,000 caught annually from 1953-1961 (Perry et al. 1999). Current estimates are a tiny fraction of former abundance. Approximately 200 fin whales have been observed in the Ross Sea (Ensor et al. 2003; Pinkerton et al. 2010).

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

Anthropogenic threats. Fin whales have undergone significant exploitation, but are currently protected under the IWC. In the Antarctic Ocean, fin whales are hunted by Japanese whalers who have been allowed to kill up to 10 fin whales each ear for the 2005-2006 and 2006-2007 seasons under an Antarctic Special Permit NMFS (2006a). Japanese whalers plan to kill 50 whales per year starting in the 2007-2008 season and continuing for the next 12 years (IWC 2006; Nishiwaki et al. 2006).

The vast majority of ship strike mortalities are never identified, and actual mortality is higher than currently documented.

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 dichlorodiphenyldichloroethylene (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).

5.3 Humpback whale

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

Southern Hemisphere. Eight proposed stocks, or populations, of humpback whales occur in waters off Antarctica (Figure 5). Individuals from these stocks winter and breed in separate areas and are known to return to the same areas. However, the degree (if any) of gene flow (i.e., adult individuals wintering in different breeding locations) is uncertain (Carvalho et al. 2011). Genetic relatedness is high between eastern and western Australian breeding populations, but significantly different (Schmitt et al. 2014). Individuals from breeding grounds in Ecuador are somewhat heterogeneous from individuals in other breeding areas, but appear to maintain a genetic linkage (Felix et al. 2009). Based on recent satellite telemetry, a revision of stocks A and G may be warranted to reflect stock movements within and between feeding areas separated east of 50º W (Dalla Rosa et al. 2008). In addition to being a breeding area, the west coast of South Africa and Namibia also appears to serve as a foraging ground due to upwelling of the Benguela Current (Barendse et al. 2010; Elwen et al. 2013). North of this, along Gabon, a separate breeding group also occurs (Elwen et al. 2013). Females appear in this area in large numbers well before their male counterparts, frequently accompanied by calves (Barendse et al. 2010). Low-level movement between breeding locations across years has been documented, bringing into question the genetic discreteness of at least Southern Hemisphere populations (particularly between Oceania groups and Australia)(Garrigue et al. 2011a; Garrigue et al. 2011b; Stevick et al. 2011). However, mixing between some populations has not been found (such as between B2 and C1 groups). Sao Tome appears to be primarily a resting, nursing, and calving area with very little breeding occurring (Carvalho et al. 2011). At least two stop over sites along Madagascar for the C stock (Fossette et al. 2014). Another breeding area may exist along the Kenya and Somali coasts, with females moving more directly along migratory corridors while males potentially searching for and intercepting females along the way (Cerchio et al. 2013). Movement between several locations, either islands or bathymetric features, in the southwestern Indian Ocean appears to be frequent (Dulau et al. 2014).

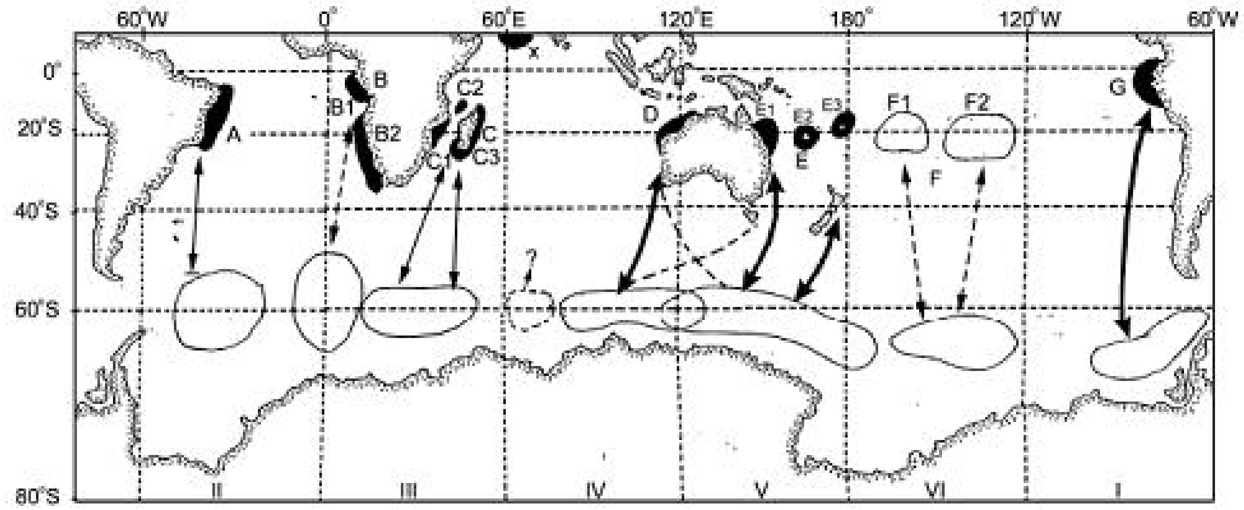

Figure 5. Southern Hemisphere humpback stocks (populations)(IWC 2005).

Distribution. Humpback whales are a cosmopolitan species that occur in the Atlantic, Indian, Pacific, and Southern oceans. Humpback whales migrate seasonally between warmer, tropical or sub-tropical waters in winter months (where they breed and give birth to calves, although feeding occasionally occurs) and cooler, temperate or sub-Arctic waters in summer months (where they feed; (Constantine et al. 2007; Garrigue et al. 2000; Gendron and Urban 1993) (Garrigue et al. 2010). The northern migration would likely be underway through the action area during the time of the proposed seismic survey and likely belong to IWC management area V and breeding stock E (Constantine et al. 2007; Dawbin 1956). In both regions, humpback whales tend to occupy shallow, coastal waters. However, migrations are undertaken through deep, pelagic waters (Winn and Reichley 1985). Some individuals may not migrate, or species occurrence in foraging areas may extend beyond summer months (Murray et al. 2014; Van Opzeeland et al. 2013).

Habitat. The east coast of the South Island and Cook Strait seem to be areas of high summer humpback whale concentration, including areas near the action area (Gibbs and Childerhouse 2000; Torres et al. 2013c). North Island occurrence near the action area there appears to be somewhat less, although sighting near the action area here have also been documented and their occurrence appears to be regular (Clement 2010). Although migration is the main activity noted, feeding in the Bay of Plenty has also been observed (Clement 2010). Additional vocal and stranding records support humpbacks as being common in New Zealand waters during summer (Berkenbush et al. 2013; McDonald 2006).

Reproduction and growth. Humpback whale calving and breeding generally occurs during winter at lower latitudes. Gestation takes about 11 months, followed by a nursing period of up to 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). Females appear to preferentially select larger-sized males (Pack et al. 2012). In some Atlantic areas, females tend to prefer shallow nearshore waters for calving and rearing, even when these areas are extensively trafficked by humans (Picanco et al. 2009). Humpback whales with calves in Hawaii also appear to prefer shallow waters, where they move more slowly than their postpartum counterparts in deeper waters who are frequently accompanied by adult males (Craig et al. 2014). Offspring appear to return to the same breeding areas at which they were born one they are independent (Baker et al. 2013).

In calving areas, males sing long complex songs directed towards females, other males, or both. The breeding season can best be described as a floating lek or male dominance polygamy (Clapham 1996). Calving occurs in the shallow coastal waters of continental shelves and oceanic islands worldwide (Perry et al. 1999). Males “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). Site fidelity off Brazilian breeding grounds was extremely low, both within and between years (Baracho-Neto et al. 2012).

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

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

Vu et al. 2012). During migration, social vocalizations are generated at 123 to 183 dB re: 1 lPa@1m with a median of 158 dB re: 1 lPa@1 m (Dunlop et al. 2013). Humpback whales increase the source level of their vocalizations in response to natural background sound level increases (Dunlop et al. 2014a).

Status and trends. Humpback whales were originally listed as endangered in 1970 (35 FR 18319), and this status remains under the ESA. Winn and Reichley (1985) argued that the global humpback whale population consisted of at least 150,000 whales in the early 1900s, mostly in the Southern Ocean. Consideration of the status of populations outside of the action area is important under the present analysis to determine the how risk the risk to the affected population(s) bears on the status of the species as a whole. Table 5 provides estimates of historic and current abundance for ocean regions.

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

Region	Population, stock, or study area	Pre-exploitation estimate	95% CI	Recent estimate	95% CI	Source
Global	~~	1,000,000	~~	~~	~~	(Roman and Palumbi 2003)
				10,000		(NMFS 1987)
Southern Hemisphere	Basinwide	100,000	~~	19,851	~~	(Gambell 1976; IWC 1996)
	Gabon	~~	~~	>1,200	~~	(Strindberg et al. 2011)
	Oceania			2,300-3,500		(Constantine et al. 2010)
	~~	~~	~~	4,329	3,345-5,313	(Constantine et al. 2012) circa 2005
	Western Australia	~~	~~	26,100[5]	20,152-33,272	(Kent et al. 2012)
	Mozambique	~~	~~	6,808	CV=0.14	(Findlay et al. 2011)
	American Samoa	~~	~~	150	~~	(Carretta et al. 2012)
	Brazil			6,404		(Andriolo et al. 2010)
	South of 60°S	~~	~~	42,000	34,000-52,000	(IWC 2007)

**Note: CIs not provided by the authors were calculated from CVs where available, using the computation from Gotelli and Ellison (2004).*

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[5] Accounting for perception bias, 33,300 Kent, C. S., C. Jenner, M. Jenner, P. Bouchet, and E. Rexstad. 2012. Southern Hemisphere Breeding Stock D humpback whale population estimates from North West Cape, Western Australia. Journal of Cetacean Research and Management 12(1):29-38.

Southern Hemisphere. The IWC recently compiled population data on humpback whales in the Southern Hemisphere. However, humpback whales in this region experienced severe whaling pressure. Based on whaling logs, particularly by Soviet vessels, at least 75,542 humpback whales were harvested from Antarctic waters from 1946 through 1973, largely from management areas IV, V, and VI (Clapham et al. 2009). One-third of these catches occurred from 1959-1961 in Area V. These numbers support Southern Hemisphere humpbacks being well below their carrying capacities (Clapham et al. 2009). A 2009 spike in calf mortality along western Australia brings into question whether carrying capacity has been reached by this population or other factors have increased mortality (Coughran and Gales 2010). Some vital rates of the humpback whale population summering off eastern Australia (E1) were recently estimated, including adult annual survival of 0.925, subadult survival of 0.70 (Hoffman et al. 2010). Growth rates for certain age classes included 10.7% for adult females and 12.4% for juveniles (Hoffman et al. 2010). Punt (2010) estimated the rate of increase for humpback whales off eastern and western Australia to be 10.9 and 10.1% annually, respectively (0.23 and 4.69 standard error, respectively). Kent et al. (2012) provided an even high estimate of 13% from 2000-2008. Humpback whales off Mozambique appear to be more numerous now than when surveyed in the 1990s (Findlay et al. 2011). Population growth of humpback whales along Brazil showed a growth rate of 7.4% annually between 1995-1998 (Ward et al. 2011).

Two separate surveys recorded 150 (Pinkerton et al. 2010) and 27 (Ensor et al. 2003) animals. It is estimated that fewer than 5% (150 animals) of the Southern Ocean population (3,000 animals) are present in the Ross Sea for only two months per year (Pinkerton et al. 2010). The current population in the Scotia Sea and Antarctic Peninsula region was estimated to be approximately 9,484 animals (Reilly et al. 2004a). However, a small number of late- or early-migrating whales may pass further south of the area during early or late austral summer, based on the species' typical migration patterns.

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

Parasites and biotoxins from red-tide blooms are other potential causes of mortality (Perry et al. 1999). The occurrence of the nematode *Crassicauda boopis* appears to increase the potential for kidney failure in humpback whales and may be preventing some populations from recovering (Lambertsen 1992). Emaciated calf and juvenile humpbacks were found in numbers an order of magnitude higher than normal along Western Australia in 2009 due to unknown causes (Coughran et al. 2013).

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 (Cole and Henry 2013). More humpback whales are killed in collisions with ships than any other whale species except fin whales (Jensen and Silber 2003).

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

Foraging locations in the North Atlantic have shifted by significant distances over the past few decades, potentially as a result of global warming (Palsboll et al. 2013).

5.4 Sei whale

Population designations. The population structure of sei whales is unknown, and populations herein assume (based on 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.

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

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

Habitat. Sei whale sightings are common in New Zealand waters, although apparently less so during austral winter months of the proposed seismic survey, including in and near the action area (Barker et al. 2009; Berkenbush et al. 2013; Clement 2010; Torres et al. 2013c). Continental shelf waters along the east coast of the North Island seem to be the area of greatest known concentration (Berkenbush et al. 2013; Clement 2010). Acoustic records also exist (Brabyn 1991). Group sizes have ranged from singletons to hundreds of individuals (Clement 2010).

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

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

Vocalization and hearing. Data on sei whale vocal behavior is limited, but includes records off the Antarctic Peninsula of broadband sounds in the 100-600 Hz range with 1.5 s duration and tonal and upsweep calls in the 200-600 Hz range of 1-3 s durations (McDonald et al. 2005). Source levels of 189 ±5.8 dB re: 1lPa 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 Hz to 44 Hz, over 1.0 s. 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 that swept from 39 Hz to 21 Hz over 1.3 s. 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

s, separated by 0.4-1.0 s) of 10-20 short (4 ms) FM sweeps between 1.5-3.5 kHz (Thomson and Richardson 1995).

Status and trends. The sei whale was originally listed as endangered in 1970 (35 FR 18319), and this status remained since the inception of the ESA in 1973. 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. The population in the Ross Sea is estimated to be around 100 animals (Pinkerton et al. 2010).

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)
Southern Hemisphere	Basinwide	63,100	--	--	--	(Mizroch et al. 1984)
	Basinwide	65,000	--	--	--	(Braham 1991)
				10,000		Boyd (2002)
	South of 60^{o}S	--	--	626	553-699	(IWC 1996)
	South of 30^{o}S	--	--	9,718	--	(IWC 1996)

**Note: CIs not provided by the authors were calculated from CVs where available, using the computation from Gotelli and Ellison (2004).*

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

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

Sei whales are occasionally killed in collisions with vessels. 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.

5.5 Southern right whale

Populations. Population structure remains uncertain, but some separation to the population level likely exists. Southern right whales breeding along Brazil and other western South Atlantic breeding areas likely belong to a single population due to the high rate of exchange within these groups (Cummings 1985; Ott et al. 2001). Genetically distinct populations also exist in waters surrounding New Zealand, southwestern Australia, and southern Africa (Baker et al. 1999; Carroll et al. 2010; Patenaude et al. 2007; Richards 2009; Valenzuela et al. 2010). Strong maternal site fidelity to breeding locations likely maintains genetic discreetness between populations (Valenzuela et al. 2010; Valenzuela et al. 2009). Some subpopulation structure appears to exist within the southwestern Atlantic population (Ott et al. 2001).

Low-level genetic exchange or movement between breeding areas appears to be occurring. Genetic exchange, at least at a low level, occurs between New Zealand/southwestern Australia whales and the southwestern Atlantic population (Moore et al. 1999; Patenaude et al. 1999; Valenzuela et al. 2010). Movement of individuals has also been documented between southwestern Atlantic breeding grounds and South African locations, although genetic exchange between these locations needs further evaluation (Best et al. 1993).

Distribution. Southern right whales occur exclusively in the southern hemisphere, currently up to 18˚ S latitude in the Atlantic and 12˚ S in the eastern Pacific (Iniguez et al. 2003; Richards 2009; Van Waerebeek et al. 1992; Van Waerebeek et al. 2009). The northern most recent sighting of a southern right whale in the western Pacific was at ~25˚ S of a mother-calf pair in Hervey Bay, Australia in 2000 and 2009 (Richards 2009). Townsend (1935) and Maury (1851; 1854) support historical distribution generally up to roughly 25-30˚ S latitude in the Pacific and Atlantic (Richards 2009). Recent sightings in New Zealand have generally been limited to between August and early November (Patenaude et al. 1999). Whaling records and surveys through the South Pacific between New Zealand and South America support a lack of southern right whale presence in this broad expanse of ocean, possibly due to lack of coastal calving habitat (Ohsumi and Kasamatsu. 1986; Richards 2009). A population appears to exist off New Zealand, with individuals from subantarctic New Zealand recolonizing former habitat along mainland New Zealand (Carroll et al. 2011a; Carroll et al. 2014b) that is distinct from another population along southeastern Australia (Carroll et al. 2011a).

Habitat. Along Peninsula Valdes, South Africa, and Brazil (all breeding or calving areas) most southern right whales remain in very shallow water of less than 20 m (Azevedo et al. 1999; Crespo et al. 2000; Elwen and Best 2004a; Elwen and Best 2004b; Payne 1986). Furthermore, right whales along South Africa prefer near shore waters that are protected from swells and wind, have gently sloping sedimentary bottoms, and lack rocky shorelines (Elwen and Best 2004a; Elwen and Best 2004b). Individuals other than cow-calf pairs tended to occupy somewhat deeper water (Elwen and Best 2004a; Elwen and Best 2004b). This near shore tendency has also been observed in the extreme north of southern right whale range along Peru (Van Waerebeek et al.

2009). Local segregation of habitat by groups may exist in which mothers and calves occupy different areas than adult male and female groups and subadult mating groups (Payne 1986). Small-scale shifts in habitat have been documented, possibly as a result of anthropogenic impacts (Rowntree et al. 2001). Port Ross along New Zealand seems to be an important calving area for right whales migrating to New Zealand (Rayment et al. 2012).

Several dozen records of southern right whales in generally nearshore waters around New Zealand have been made over the past several decades, primarily during the same season as the proposed seismic survey (Berkenbush et al. 2013; Patenaude 2003a; Richards 2002a). An area along the northern shore of the North Island appears to be a significant calving ground (Patenaude 2003a). However, at least one sighting has been made near the offshore region of the proposed action area (Torres et al. 2013a). As supported by observations in New Zealand and in other areas around the world, habitat modelling in New Zealand suggests that offshore habitat is a less likely place for southern right whales to occur during this time than in sheltered nearshore areas (Torres et al. 2013a; Torres et al. 2013b; Torres et al. 2013c; Torres et al. 2013d).

Migration and movement. Southern right whales migrate between winter breeding areas in coastal waters of the South Atlantic, Pacific, and Indian Oceans from May to December and offshore summer (January to April) foraging locations in the Subtropical and Antarctic Convergence zones (Azevedo et al. 1999; Bannister et al. 1999; De Oliveira et al. 2009; Tormosov et al. 1998). Movements are not necessarily north-south, but may also be east-west, such as between South African breeding grounds and Gough Island (south-central South Atlantic Ocean)(Best et al. 1993; De Oliveira et al. 2009; Mate et al. 2011). Females with calves stay significantly longer (~71 days) in calving grounds off southern Australia than do females without calves (~20 days)(Burnell and Bryden 1997). Southern right whales have been seen in these waters from mid-May to late October, with 100% of calves being born by September/October (Burnell and Bryden 1997). Southern right whales appear in waters off New Zealand’s South Island in May, likely for calving (Richards 2002a). Clement (2010) suggested that East Cape may be a point at which southern right whales concentrate along their migration route. Females with calves born late in the season stayed twice as long as those whose calves were born early in the season (80 versus 40 days)(Burnell and Bryden 1997).

Reproduction. Breeding areas are known in the southwestern Atlantic, off southern Africa (South Africa, extending along Namibia), and in Oceania (southwestern Australia as one population, New Zealand as another), where individuals gather in the austral winter (Iniguez et al. 2003; Roux et al. 2001; Valenzuela et al. 2010). Reproductive females appear to migrate to breeding/calving grounds in years when they give birth, but are seen less frequently in other years (Payne 1986). Whales arrive in late June along Peninsula Valdes, breeding appears to peak in August, calving appears to peak around September, and whales disappear in November (Crespo et al. 2000; Cummings et al. 1972b). Similar sighting trends are evident further north along southeastern Brazil (De Oliveira Santos et al. 2001). Waters off southern Australia and the

Auckland Islands also serve as a calving region (Barrett et al. 1999; Burnell and Bryden 1997). Adult female survival is <97% annually off South Africa and first year survival of calves is 71.3-91.3% (Best et al. 2001; Best and Kishino 1998; Brandao et al. 2010; Cooke et al. 2001).

Best et al. (2001), Brandao et al. (2010), and Cooke et al. (2001) estimated the age at first parturition to be 7-9 years in southwestern Atlantic and South Africa breeding areas. Soviet whaling data suggest a pregnancy rate of 28.1% (Tormosov et al. 1998). Gestation is estimated at between 357 and 396 days (Best 1994). Calves are born at sizes as small as 4.53 m (Best and Ruther 1992). Calves born to first-time mothers are smaller than those born to more experienced females off South Africa and suffer a higher mortality rate as a result (Best and Ruther 1992; Elwen and Best. 2004). Growth rates based upon aerial photography indicate calf growth rates of 2.8 cm per day in the same area (Best and Ruther 1992). Adult females may growth 50 cm longer than males (Tormosov et al. 1998).

Behavior. The largest group size observed along Australia has been 16 individuals (Kemper et al. 1997). Shortly after parturition, southern right whale females engage in rapid travel with their calves; this activity is theorized to develop the calves' swimming and surfacing skills and deter predator detection (Thomas and Taber. 1984). After three to four weeks, travel slows while calf resting and play behavior increases, the later centered around the mother (Thomas and Taber. 1984). Calves remain close to their mothers over 90% of the time during the first few weeks of life, although separations become more frequent and longer as the calf grows older (Taber and Thomas. 1982). Days before migration south, traveling activity again increases (Thomas and Taber. 1984). Separation between mother and calf also decreases just before migration (Taber and Thomas. 1982). Nursing comprises roughly 5% of a calves' time budget in a given 24-hour period, although nursing bout duration increases as the calf grows larger (Thomas and Taber. 1984). Yearlings may or may not return the following year with their mothers (Taber and Thomas. 1982; Thomas and Taber. 1984). Interestingly, 93% of yearlings that return to breeding grounds with their mothers are female (Best, 2003 #188}. The incidence of yearlings with their mothers dropped sharply at one year of age; yearlings likely yean at this age, if not sooner (Best et al. 2003).

Southern right whales appear to be generally solitary or form dyads when not on breeding grounds, but form groups when on northern breeding areas (Best et al. 2003; Ohsumi and Kasamatsu. 1986). If not accompanied by yearlings or calves, most individuals participate in surface active groups (composed mostly of males centered around a pre-pubescent focal female; 2-20 individuals, average of 3-5, and number of individuals increasing through late August and September, likely a function of decreasing female availability) or engaged in courtship or non-active groups (composed mostly of females; 2-5 individuals, average of 2.27)(Best et al. 2003; Costa et al. 2007). Spyhopping is a common behavior while on breeding and calving grounds (Costa et al. 2007). Most conceptions are believed to occur within a four month period centered around Mid-July (Best et al. 2003).

Acoustics and hearing. Southern right whales are known to produce a variety of low-frequency vocalizations on breeding groups (Clark 1982a; Clark 1983). The most common is an “up” call (50-200 Hz with a frequency modulated upsweep)(Tellechea and Norbis 2012) that appears to establish contact and aggregate individuals (Clark 1982a; Clark 1983; Dunshea and Gedamke. 2010). Blows (100-400 Hz), apart from their respiratory function, may signal calves to remain close or inform approaching animals to stay away (particularly loud, pulsative, or tonal blows)(Clark 1981; Clark 1982b; Clark 1983). Slaps (50-1,000 Hz), “pulsatile” and “hybrid” (both complex 50-200 Hz) calls may have a threat, antagonism, or aggressive function in sexually active groups, but little or no communicative function in other groups (Clark 1981; Clark 1982b; Clark 1983; Tellechea and Norbis 2012). “Down” calls (100-200 Hz with frequency modulated downsweep) seem to have a similar function as “up” calls, but are produced by more excited individuals (Clark 1981; Clark 1982b; Clark 1983). “High” calls (200-500 Hz) may indicate an excited individual, excite other individuals, or attract whales to a group (Clark 1981; Clark 1982b; Clark 1983; Tellechea and Norbis 2012). Resting and swimming groups are silent most of the time; resting groups do make “blow” sound, though, and resting individuals may make “up” calls (Clark 1981; Clark 1983). Swimming groups make “up” calls more than any other sound (Clark 1981; Clark 1983). As physical activity increases, so does the level of vocal activity (Clark 1981; Clark 1983). Mildly active groups were silent only one-quarter of the time and made large numbers of “slap” sounds by striking flukes or flippers on the water surface (Clark 1981; Clark 1983). Highly-active groups are rarely silent and make numerous slaps, blows, as well as, “up”, “hybrid”, “high”, and “pulsative” calls (Clark 1981; Clark 1983). Sexually active groups have not been documented to make “up” calls or be silent, but make extensive “hybrid”, “high”, and “pulsative” calls (Clark 1981; Clark 1983). “Up” calls have also been recorded during winter on feeding grounds (Sirovic et al. 2006).

Status and trends. Southern right whale populations in general appear to be increasing at a robust rate. De Oliveira (2009) estimated that roughly 7,000 individuals exist today; 5-10% of the species former abundance. The Australian recovery plan for southern right whales estimates that 60,000 southern right whales existed prior to commercial whaling; 1,500 individuals are estimated to visit waters around Australia (NHT 2005). Population growth off Australia is believed to be 7-13% annually (Bannister 2001). New Zealand has estimated that 16,000 individuals visited its waters prior to commercial exploitation; this number was believed to have been reduced to between 14-52 individuals and current abundance is less than 5% of historic levels (Patenaude 2003b). Genetic analyses suggest a reproductive male population of 1,001 individuals for New Zealand southern right whales (Carroll et al. 2012). IWC (2014) estimated 12,000 southern right whales.

Some evidence suggests that, due to historical shore whaling, southern right whales have not reoccupied former calving or nursery habitat, particularly in northern areas of the species’ historical range (Kemper and Samson. 1999; Richards 2009). Whaling records indicate that southern right whales formerly inhabited waters much further north than presently known relatively late in spring, such as the Kermadec Islands (29˚ S) 1,000 km to the north-northeast of New Zealand (Richards 2002b). Southern right whales were not sighted at all in New Zealand from 1927-1963, but roughly 70 sightings (30-50 individuals) per year have been documented since 2003 (Richards 2009).

Genetic diversity of populations may also have been reduced as a result of extensive whaling, although this is not the case for all populations (Baker et al. 1999; Valenzuela et al. 2010).

Southern right whales regularly winter in Campbell Island south of New Zealand and have been seen with increasing frequency along the mainland (Carroll et al. 2011a; Carroll et al. 2011b; Childerhouse et al. 2010; Patenaude 2003a; Patenaude and Baker 2001; Patenaude et al. 2001; Stewart and Todd 2001). Their numbers seem to be increasing and the latest estimate of population size is 2,169 (Carroll et al. 2013).

Natural threats. The only natural predator of southern right whales are killer whales (Sironi et al. 2008). In some cases, such as off Peninsula Valdes, Argentina, southern right whales appear to have abandoned habitat where the highest concentrations of killer whales also occur (Sironi et al. 2008). When attacked, southern right whales attempt to turn their tails towards attacking killer whales and use their flukes as a weapon (Ford and Reeves. 2008; Sironi et al. 2008). Right whale females will also attempt to protect their calves, move into shallower water, increase group size, and form a rosette formation (circle with tails facing outward, younger individuals in the center; this activity was formerly only known for sperm whales being attacked by killer whales)(Ford and Reeves. 2008; Sironi et al. 2008). Southern right whale aggregations in calving areas have been suggested to be a result of predator defense (Ford and Reeves. 2008; Sironi et al. 2008).

Anthropogenic threats. Southern right whales underwent severe decline due to whaling during the 18th and 19th centuries (Costa et al. 2005; NHT 2005). At least 6,262 individuals were killed when Townsend (1935) published a summary of whaling records (Richards 2009). However, these numbers may be much higher, as Richards (2009) suggests roughly 20,000 individuals had been killed around New Zealand by 1927 alone. Illegal Soviet whaling removed at least an additional 3,368 individuals between 1951 and 1971 (Richards 2009; Tormosov et al. 1998). Some 53,000 to 58,000 were likely taken from waters along New Zealand and eastern Australia (Carroll et al. 2014a).

Southern right whales are currently subject to many of the same anthropogenic threats as other large whales face. In the Southern Hemisphere, southern right whales are by far the most ship struck cetacean, with at least 56 reported instances; nearly four-fold higher than the second most struck large whale (Van Waerebeek et al. 2007). Over the past ~68 years in Australia, one whale was documented to have been shot (non-fatal), one fatal and 12 non-fatal entanglements (most frequently by lines, nets, and buoys) have been documented, and three non-fatal and two fatal ship strikes have been documented out of 44 mortalities or non-fatal anthropogenic interactions (Kemper et al. 2008). Two-thirds of these events occurred from July to October, but occurrences in every month except January are known (Kemper et al. 2008). The incidence of human interactions of this type has increased four-fold since the mid-1970s (Kemper et al. 2008). Overall, 11% of records for Australia involve ship strike versus 16% in South Africa and 35% for North Atlantic right whales (Kemper et al. 2008). Additional threats identified in Australian waters include water quality and pollution and near shore habitat degradation due to development (NHT 2005).

5.6 Sperm whale

Populations. There is no clear understanding of the global population structure of sperm whales (Dufault et al. 1999). Recent ocean-wide genetic studies indicate low, but statistically significant, genetic diversity and no clear geographic structure, but strong differentiation between social groups (Lyrholm and Gyllensten 1998; Lyrholm et al. 1996; Lyrholm et al. 1999). Chemical analysis also suggests significant differences in diet for animals captured in different regions of the North Atlantic. However, vocal dialects indicate parent-offspring transmission that support differentiation in populations (Rendell et al. 2011). Vocal differences exist not only across ocean basins, but also over much smaller spatial scales (Amano et al. 2014). Therefore, population-level differences may be more extensive than are currently understood.

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

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

Movement. Mature males range between 70º N in the North Atlantic and 70º S in the Southern Ocean (Perry et al. 1999; Reeves and Whitehead 1997), whereas mature females and immature individuals of both sexes are seldom found higher than 50º N or S (Reeves and Whitehead 1997). In winter, sperm whales migrate closer to equatorial waters (Kasuya and Miyashita 1988; Waring et al. 1993) where adult males join them to breed. Movement patterns of Pacific female and immature male groups appear to follow prey distribution and, although not random, movements are difficult to anticipate and are likely associated with feeding success, perception of the environment, and memory of optimal foraging areas (Whitehead et al. 2008). However, no sperm whale in the Pacific has been known to travel to points over 5,000 km apart and only rarely have been known to move over 4,000 km within a time frame of several years. This means that although sperm whales do not appear to cross from eastern to western sides of the Pacific (or vice-versa), significant mixing occurs that can maintain genetic exchange. Movements of several hundred kilometers are common, (i.e. between the Galapagos Islands and the Pacific coastal Americas). Movements appear to be group or clan specific, with some groups traveling straighter courses than others over the course of several days. However, general transit speed averages about 4 km/h. Sperm whales in the Caribbean region appear to be much more restricted in their movements, with individuals repeatedly sighted within less than 160 km of previous sightings.

Habitat. Sperm whales have a strong preference for waters deeper than 1,000 m (Reeves and Whitehead 1997; Watkins 1977), although Berzin (1971) reported that they are restricted to waters deeper than 300 m. While deep water is their typical habitat, sperm whales are rarely found in waters less than 300 m in depth (Clarke 1956; Rice 1989b). 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 may also be temporary feeding areas for sperm whales (Griffin 1999; Jaquet et al. 1996; Waring et al. 1993).

Sperm whales occur in both nearshore and offshore waters year-round along New Zealand (Berkenbush et al. 2013; Torres et al. 2013b). Hawke's Bay may represent a migratory pass through, but resident sperm whales are found for weeks or months continuously near Kaikoura (Clement 2010; Jaquet et al. 2000; Lettevall et al. 2002; Richter et al. 2003a; Sagnol et al. 2014). Fine-scale distribution data support occurrence further offshore during summer in Kaikoura versus in the winter (where they are waters of 500-1,000 m deep) (Jaquet et al. 2000; Richter et al. 2003a). Hundreds of sperm whales have stranded along New Zealand's shores (Berkenbush et al. 2013; Brabyn 1991).

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 every four to six years between the ages of 12 and 40 (Kasuya 1991; Whitehead et al. 2008). It has been suggested that some mature males may not migrate to breeding grounds annually during winter, and instead may remain in higher latitude feeding grounds for more than one year at a time (Whitehead and Arnbom 1987).

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

Stable, long-term associations among females form the core of sperm whale societies (Christal et al. 1998). Up to about a dozen females usually live in such groups, accompanied by their female and young male offspring. Young individuals are subject to alloparental care by members of either sex and may be suckled by non-maternal individuals (Gero et al. 2009). Group sizes may be smaller overall in the Caribbean Sea (6-12 individuals; 7-9 along Dominica) versus the Pacific (25-30 individuals)(Gero et al. 2013; Jaquet and Gendron 2009). Groups may be stable for long periods, such as for 80 days in the Gulf of California (Jaquet and Gendron 2009). Males start leaving these family groups at about 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).

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

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

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

Status and trends. Sperm whales were originally listed as endangered in 1970 (35 FR 18319), and this status remained with the inception of the ESA in 1973. Although population structure of sperm whales is unknown, several studies and estimates of abundance are available. Consideration of the status of populations outside of the action area is important under the present analysis to determine the how risk the risk to the affected population(s) bears on the status of the species as a whole. Table 7 contains historic and current estimates of sperm whales 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).

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

Region	Population, stock, or study area	Pre-exploitation estimate	95% CI	Recent estimate	95% CI	Source
Global	~~	~~	~~	900,000	~~	(Würsig et al. 2000)
	~~	1,110,000	672,000-1,512,000	360,000	105,984-614,016*	(Whitehead 2002)
Southern Hemisphere	Basinwide	547,600	~~	299,400	~~	(Gosho et al. 1984; IWC 1988; Perry et al. 1999)
				30,000		Boyd (2002)
	South of 60°S	~~	~~	14,000	8,786-19,214*	(Butterworth et al. 1995) as cited in (Perry et al. 1999)
	South of 60°S	~~	~~	12,069	~~	(Whitehead 2002b)
	South of 30°S	~~	~~	128,000	17,613-238,387*	(Butterworth et al. 1995) as cited in (Perry et al. 1999)

**Note: CIs not provided by the authors were calculated from CVs where available, using the computation from Gotelli and Ellison (2004).*

Southern Hemisphere. Whaling in the Southern Hemisphere averaged roughly 20,000 whales between 1956-1976 (Perry et al. 1999). Population size appears to be stable (Whitehead 2003). Populations of sperm whales in the Ross Sea are estimated to range between 88 (Ensor et al. 2003) and 800 (Pinkerton et al. 2010) animals.

Natural threats. Sperm whales are known to be occasionally preyed on 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 1989a; Weller et al. 1996; Whitehead 1995). Strandings are also relatively common events, with one to dozens of individuals generally beaching themselves and dying during any single event. Although several hypotheses, such as navigation errors, illness, and anthropogenic stressors, have been proposed (Goold et al. 2002; Wright 2005), direct widespread causes of strandings remain unclear. Calcivirus and papillomavirus are known pathogens of this species (Lambertsen et al. 1987; Smith and Latham 1978).

Anthropogenic threats. Sperm whales historically faced severe depletion from commercial whaling operations. From 1800 to 1900, the IWC estimated that nearly 250,000 sperm whales were killed by whalers, with another 700,000 from 1910 to 1982 (IWC Statistics 1959-1983). However, other estimates have included 436,000 individuals killed between 1800-1987 (Carretta et al. 2005). However, all of these estimates are likely underestimates due to illegal and inaccurate killings by Soviet whaling fleets between 1947-1973. In the Southern Hemisphere, these whalers killed an estimated 100,000 whales that they did not report to the IWC (Yablokov et al. 1998). 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.

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

Contaminants have been identified in sperm whales, but vary widely in concentration based on 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.

Small changes in reproductive parameters, such as the loss of adult females, can significantly alter the population trajectory of sperm whale populations (Chiquet et al. 2013).

5.7 Green sea turtle

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

Based upon genetic differences, two or three distinct regional clades may exist in the Pacific: western Pacific and South Pacific islands, eastern Pacific, and central Pacific, including the rookery at French Frigate Shoals, Hawaii (Dutton and Balazs ; Dutton et al. 1996).

Table 8. Locations and most recent abundance estimates of threatened green sea turtles as annual nesting females (AF).

Location	Most recent abundance	Reference
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)

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

Growth and reproduction. Most green sea turtles exhibit particularly slow growth rates, which have been attributed to their largely plant-eating diet (Bjorndal 1982). Growth rates of juveniles vary substantially among populations, ranging from <1 cm/year (Green 1993) to >5 cm/year (McDonald Dutton and Dutton 1998), likely due to differences in diet quality, duration of foraging season (Chaloupka et al. 2004), and density of turtles in foraging areas (Balazs and Chaloupka 2004; Bjorndal et al. 2000; Seminoff et al. 2002b). Hart et al. (2013a) found growth rates of green sea turtles in the U.S. Virgin Islands to range from 0-9.5 cm annually (mean of 4.1, SD 2.4). The largest growth rates were in the 30-39 cm class. If individuals do not feed sufficiently, growth is stunted and apparently does not compensate even when greater-than-needed resources are available (Roark et al. 2009). In general, there is a tendency for green sea turtles to exhibit monotonic growth (declining growth rate with size) in the Atlantic and non-monotonic growth (growth spurt in mid-size classes) in the Pacific, although this is not always the case (Balazs and Chaloupka 2004; Chaloupka and Musick 1997; Seminoff et al. 2002b). It is estimated that green sea turtles reach a maximum size just under 100 cm in carapace length (Tanaka 2009). A female-bias has been identified from studies of green sea turtles (Wibbels 2003).

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

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

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

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

2010a). East Pacific adults migrate along coastal corridors between Central American nesting and foraging locations (Blanco et al. 2012).

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

Green sea turtles in the Gulf of Mexico tend to remain along the coast (lagoons, channels, inlets, and bays), with nesting primarily occurring in Florida and Mexico and infrequent nesting in all other areas (Landry and Costa 1999; Meylan et al. 1995; NMFS and USFWS 1991; USAF 1996). Foraging areas seem to be based upon seagrass and macroalgae abundance, such as in the Laguna Madre of Texas. However, green sea turtles may also occur in offshore regions, particularly during migration and development. Sea turtle frequently forage far from their nesting beaches. Sea turtles foraging in the western Gulf of Mexico almost exclusively stem from Gulf of Mexico and northern Caribbean rookeries (Anderson et al. 2013).

Green sea turtles are the second most frequently reported sea turtle species behind leatherbacks in New Zealand (Gill 1997a). Green sea turtles occur along New Zealand shores and have been recorded in and near the seismic survey action area (Gill 1997a; NZDOC 2014a; NZDOC 2014b). Frequencies would likely relatively low during the time of the seismic survey (Gill 1997a). Individuals present are likely from the Indo-Pacific or eastern Pacific populations (NZDOC 2014b). Rare instances of bycatch are also documented (Harley and Kendrick 2006).

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

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

These hearing sensitivities are similar to those reported for two terrestrial species: pond and wood turtles. Pond turtles are 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 3000 Hz (Wever and Vernon 1956). Wood turtles are sensitive up to about 500 Hz, followed by a rapid decline above 1000 Hz and almost no responses beyond 3000 or 4000 Hz (Patterson 1966).

Status and trends. Federal listing of the green sea turtle occurred on July 28, 1978, with all populations listed as threatened except for the Florida and Pacific coast of Mexico breeding populations, which are endangered (43 FR 32800). 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 12 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).

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

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

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

Anthropogenic threats. Major anthropogenic impacts to the nesting and marine environment affect green sea turtle survival and recovery (Patino-Martinez 2013). At nesting beaches, green sea turtles rely on intact dune structures, native vegetation, and normal beach temperatures for nesting (Ackerman 1997). Structural impacts to nesting habitat include the construction of buildings and pilings, beach armoring and renourishment, and sand extraction (Bouchard et al. 1998; Lutcavage et al. 1997). These factors may directly, through loss of beach habitat, or indirectly, through changing thermal profiles and increasing erosion, serve to decrease the amount of nesting area available to nesting females, and may evoke a change in the natural behaviors of adults and hatchlings (Ackerman 1997; Witherington et al. 2003; Witherington et al. 2007). On the Pacific coast of Mexico in the mid-1970s, >70,000 green turtle eggs were harvested every night. Hundreds of mostly immature green sea turtles were killed between 2006 and 2008 due to bycatch and direct harvest along Baja California Sur (Senko et al. 2014). The presence of lights on or adjacent to nesting beaches alters the behavior of nesting adults (Witherington 1992) and is often fatal to emerging hatchlings as they are attracted to light sources and drawn away from the water (Witherington and Bjorndal 1991). In addition to impacting the terrestrial zone, anthropogenic disturbances also threaten coastal marine habitats, particularly areas rich in seagrass and marine algae. These impacts include contamination from herbicides, pesticides, oil spills, and other chemicals, as well as structural degradation from excessive boat anchoring and dredging (Francour et al. 1999; Lee Long et al. 2000; Waycott et al. 2005). Ingestion of plastic and other marine debris is another source of morbidity and mortality (Stamper et al. 2009). Green sea turtles stranded in Brazil were all found to have ingested plastics or fishing debris (n=34), although mortality appears to have 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). However, a legal fishery operates in Madagascar that harvested about 10,000 green turtles annually in the mid-1990s. Green sea turtles are killed because they are seen as competitors for fishery resources in parts of India (Arthur et al. 2013). Between 1991 and 2011, an average of 8,169 green sea turtles were harvested annually along the Caribbean coast of Nicaragua (over 171,000 over this period); a rate that has been in decline potentially due to population depletion (Lagueux et al. 2014). Between 16 and 20% of green sea turtles in a Galapagos port (a foraging area) displayed evidence of ship strike; this dropped to about 3% in nesting areas (Denkinger et al. 2013). Entrapment or incapacitation due to beach debris is responsible for 0.3-1.0% of nesting females on a nesting beach in Seychelles; an issue that may be problematic in other nesting locations (Mortimer and von Brandis 2013).

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

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), with similar levels in both sexes (da Silva et al. 2014). 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).

5.8 Hawksbill sea turtle

Populations. Populations are distinguished generally by ocean basin and more specifically by nesting location. Our understanding of population structure is relatively poor.

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

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

Habitat. Hawksbill sea turtles are highly migratory and use a wide range of broadly separated localities and habitats during their lifetimes (Musick and Limpus 1997; Plotkin 2003). Small juvenile hawksbills (5-21 cm straight carapace length) have been found in association with *Sargassum* spp. in both the Atlantic and Pacific oceans (Musick and Limpus 1997) and observations of newly hatched hawksbills attracted to floating weed have been made (Hornell 1927; Mellgren and Mann 1996; Mellgren et al. 1994). Post-oceanic hawksbills may occupy a range of habitats that include coral reefs or other hard-bottom habitats, sea grass, algal beds, mangrove bays and creeks (Bjorndal and Bolten 2010; Musick and Limpus 1997), and mud flats (R. von Brandis, unpublished data in NMFS and USFWS 2007c). Eastern Pacific adult females have recently been tracked in saltwater mangrove forests along El Salvador and Honduras, a habitat that this species was not previously known to occupy (Gaos et al. 2011). Individuals of multiple breeding locations can occupy the same foraging habitat (Bass 1999; Bowen et al. 1996; Bowen et al. 2007; Diaz-Fernandez et al. 1999; Velez-Zuazo et al. 2008). As larger juveniles, some individuals may associate with the same feeding locality for more than a decade, while others apparently migrate from one site to another (Blumenthal et al. 2009; Mortimer et al. 2003; Musick and Limpus 1997). Larger individuals may prefer deeper habitats than their smaller counterparts (Blumenthal et al. 2009). Nesting sites appear to be related to beaches with relatively high exposure to wind or wind-generated waves (Santana Garcon et al. 2010).

Hawksbill sea turtles are rare in New Zealand mainland waters, with roughly three dozen total records, including in the proposed action area (Gill 1997a; NZDOC 2014a; NZDOC 2014b). Austral winter records are the most numerous, but records during the time of the proposed seismic survey are also available (Gill 1997a).

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

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

Hatchlings in Brazil exhibit a strong female bias of 89-96% (dei Marcovaldi et al. 2014).

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

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 1994a; Lenhardt 2002; Moein Bartol and Ketten 2006; Ridgway et al. 1969b). Piniak et al. (2012) found hawksbill hatchlings capable of hearing underwater sounds at frequencies of 50-1,600 Hz (maximum sensitivity at 200-400 Hz). Hearing below 80 Hz is less sensitive but still possible (Lenhardt 1994a).

These hearing sensitivities are similar to those reported for two terrestrial species: pond and wood turtles. Pond turtles are 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 3000 Hz (Wever and Vernon 1956). Wood turtles are sensitive up to about 500 Hz, followed by a rapid decline above 1000 Hz and almost no responses beyond 3000 or 4000 Hz (Patterson 1966).

Status and trends. Hawksbill sea turtles received protection on June 2, 1970 (35 FR 8495) under the Endangered Species Conservation Act and since 1973 have been listed as endangered under the ESA. Although no historical records of abundance are known, hawksbill sea turtles are considered to be severely depleted due to the fragmentation and low use of current nesting beaches (NMFS and USFWS 2007c). Worldwide, an estimated 21,212-28,138 hawksbills nest each year among 83 sites. Among the 58 sites for with historic trends, all show a decline during the past 20 to 100 years. Among 42 sites for which recent trend data are available, 10 (24%) are increasing, three (7%) are stable and 29 (69%) are decreasing.

Pacific Ocean. American Samoa and Western Samoa host fewer than 30 females annually (Grant et al. 1997; Tuato'o-Bartley et al. 1993). In Guam, only 5-10 females are estimated to nest annually (G. Balazs, NMFS, in litt. to J. Mortimer 2007; G. Davis, NMFS, in litt. to J. Mortimer 2007) and the same is true for Hawaii, but there are indications that this population is increasing (G. Balazs, pers. comm. in NMFS and USFWS 2007c). Additional populations are known from the eastern Pacific (potentially extending from Mexico through Panama), northeastern Australia, and Malaysia (Hutchinson and Dutton 2007). Total number of nesting females for the Central Pacific hawksbill population was estimated at 940 – 1,200 females annually for the last few years, with an overall downward trend (NMFS and USFWS 2007a).

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

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

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

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

5.9 Leatherback sea turtle

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

Pacific Ocean. In the Pacific Ocean, genetic studies have identified three distinct populations (referred to also as genetic stocks or Management Units; see Wallace et al. 2010b) of leatherback turtles: (1) Mexico and Costa Rica, which are genetically homogenous but distinct from the western populations; (2) Papua Barat in Indonesia, Papua New Guinea, Solomon Islands, and Vanuatu, which comprise a metapopulation representing a single genetic stock; and (3) Malaysia (Barragan and Dutton 2000; Dutton et al. 1999; Dutton 2005-2006; Dutton et al. 2000a; Dutton et al. 2006; Dutton 2007). The genetically distinct Malaysia nesting population likely is extirpated (Chan and Liew 1996b; Dutton et al. 1999; Dutton 2005-2006).

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 1997b; Hodge and Wing 2000).

Growth and reproduction. It has been thought that leatherbacks reach sexual maturity somewhat faster than other sea turtles (except Kemp’s ridley), with an estimated range of 3-6 years (Rhodin 1985) to 13-14 years (Zug and Parham 1996). However, recent research suggests otherwise, with western North Atlantic leatherbacks possibly not maturing until as late as 29 years of age (Avens and Goshe 2007; Avens and Goshe 2008; Avens et al. 2009). Female leatherbacks nest frequently (up to 13, average of 5-7 nests per year and about every 2-3 years)(Eckert et al. 2012). The average number of eggs per clutch varies by region: Atlantic Ocean (85 eggs), western Pacific Ocean (85 eggs), eastern Pacific Ocean (65 eggs) and Indian Ocean (>100 eggs(Eckert et al. 2012). However, up to ~30% of the eggs can be infertile. Thus, the actual proportion of eggs that can result in hatchlings is less than this seasonal estimate. The eggs incubate for 55-75 days before hatching.

Habitat. Leatherbacks occur throughout marine waters, from nearshore habitats to oceanic environments (Grant and Ferrell 1993; Schroeder and Thompson 1987; Shoop and Kenney 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). Cool, shallow, productive waters are areas where leatherbacks concentrate during late fall, winter, and early spring, where their dives become shallower and shorter, presumably associated with foraging opportunities (Dodge et al. 2014). 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).

For the western Pacific population, seven ecoregions (South China/Sulu and Sulawesi Seas, Indonesian Seas, East Australian Current Extension, Tasman Front, Kuroshio Extension, equatorial Eastern Pacific, and California Current Extension) were identified as important seasonal foraging areas (Benson et al. 2011a).

Leatherback sea turtles are the most common sea turtles in New Zealand waters, with over 120 records (Gill 1997a; NZDOC 2014a; NZDOC 2014b). The North Island appears to be a foraging location (NZDOC 2014b). Most records are from the austral summer, but instances from May and June are also known (Gill 1997a). Bycatch in deep, offshore waters has also occurred, mostly along the North Island (Harley and Kendrick 2006). One Individual satellite tagged in Papua New Guinea appeared to migrate south to waters east of New Zealand before moving north again (Benson et al. 2007b).

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. 2009).

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

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) and temperatures above 30C increase mortality (Santidrián Tomillo et al. 2014).

Feeding. Leatherbacks may forage in high-invertebrate prey density areas formed by favorable features (Eckert 2006; Ferraroli et al. 2004). Although leatherbacks forage in coastal waters, they appear to remain primarily pelagic through all life stages (Heppell et al. 2003). The location and abundance of prey, including medusae, siphonophores, and salpae, in temperate and boreal latitudes likely has a strong influence on leatherback distribution in these areas (Plotkin 1995). Leatherback prey are frequently found in the deep-scattering layer in the Gulf of Alaska (Hodge and Wing 2000). North Pacific foraging grounds contain individuals from both eastern and western Pacific rookeries, although leatherbacks from the eastern Pacific generally forage in the Southern Hemisphere along Peru and Chile (Dutton 2005-2006; Dutton et al. 2000b; Dutton et al. 1998). Mean primary productivity in all foraging areas of western Atlantic females is 150% greater than in eastern Pacific waters, likely resulting in twice the reproductive output of eastern Pacific females (Saba et al. 2007). Leatherbacks have been observed feeding on jellyfish in waters off Washington State and Oregon (Eisenberg and Frazier 1983; Stinson 1984a).

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 1994a; 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 1994a). Sounds have also been recorded during hatchlings that may help to coordinate emergence, generally ranging from about 100 to 1,700 Hz, but some up to 24 kHz (Ferrara et al. 2014).

These hearing sensitivities are similar to those reported for two terrestrial species: pond and wood turtles. Pond turtles are 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 3000 Hz (Wever and Vernon 1956). Wood turtles are sensitive up to about 500 Hz, followed by a rapid decline above 1000 Hz and almost no responses beyond 3000 or 4000 Hz (Patterson 1966).

Status and trends. Leatherback sea turtles received protection on June 2, 1970 (35 FR 8491) under the Endangered Species Conservation Act and, since 1973, have been listed as endangered under the ESA, but declines in nesting have continued worldwide. Consideration of the status of populations outside of the action area is important under the present analysis to determine the 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 1982b). 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).

Heavy declines have occurred at all major Pacific basin rookeries, as well as Mexico, Costa Rica, Malaysia, India, Sri Lanka, Thailand, Trinidad, Tobago, and Papua New Guinea (Table 9). This includes a nesting decline of 23% between 1984-1996 at Mexiquillo, Michoacán, Mexico (Sarti et al. 1996). Only an Indonesian nesting assemblage has remained relatively abundant in the Pacific basin. The largest extant leatherback nesting assemblage in the Indo-Pacific lies on the northern Vogelkop coast of Irian Jaya (West Papua), Indonesia, with roughly 3,000 nests recorded annually (Dutton et al. 2007; Putrawidjaja 2000; Suárez et al. 2000). The Western Pacific leatherback metapopulation harbors the last remaining nesting aggregation of significant size in the Pacific with approximately 2700–4500 breeding females (Dutton et al. 2007; Hitipeuw et al. 2007). The total number of nests per year for the Jamursba-Medi leatherback nesting population ranged between a high of 6,373 nests in 1996 and a low of 1,537 nests in 2010 (Hitipeuw et al. 2007) and 1,596 in 2011 (Tapilatu et al. 2013). Nesting at Terengganu, Malaysia is 1% of that in 1950s (Chan and Liew 1996a). The South China Sea and East Pacific nesting colonies have undergone catastrophic collapse. Overall, Pacific populations have declined from an estimated 81,000 individuals to <3,000 total adults and subadults (Spotila et al. 2000). The number of nesting leatherbacks has declined by an estimated 95% over the past 20 years in the Pacific (Gilman 2009). Drastic overharvesting of eggs and mortality from fishing activities is likely responsible for this tremendous decline (Eckert 1997; Sarti et al. 1996). The current overall estimate for Papua Barat, Indonesia, Papua New Guinea, and Solomon Islands is 5,000 to 10,000 nests per year (Nel 2012).

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

Table 9. Leatherback nesting population site location information where multiple-year surveys were conducted or trends are known (data type, years surveyed, annual number (nests, females, trend).

Location	Data: Nests, Females	Years	Annual number	Trend	Reference
Pacific					
Indonesia (Papua-Jamursba-Medi)	Nests	1984 - 2011	14,522-1,596	▼	Tapilatu et al. (2013)
Indonesia (Papua-Wermon)	Nests	2002 - 2011	2,994-1,096	▼	Tapilatu et al. (2013)
Papua New Guinea (Labu Tali)	Nests	1989 - 2011	76–59	▼[4]	Hirth et al. (1993); Pilcher 2011a
Vanuatu	Nests	2002 - 2010	~50	▼	Petro 2011; Petro et al. (2007)
Malaysia (Terengganu)	Nests	1956 - 2009	10,000-10	▼	reviewed by Eckert et al. (2012)

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

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

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

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

Natural threats. Sea turtles face predation primarily by sharks and to a lesser extent by killer whales (Pitman and Dutton 2004). Hatchlings are preyed upon by herons, gulls, dogfish, and sharks. Leatherback hatching success is particularly sensitive to nesting site selection, as nests that are overwashed have significantly lower hatching success and leatherbacks nest closer to the high-tide line than other sea turtle species (Caut et al. 2009b). The fungal pathogens *Fusarium falciforme* and *F. keratoplasticum* 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). Lost timber on beaches and beachcast logs can trap adult females nesting on Gabon beaches, resulting in mortality (Ikaran 2013).

Anthropogenic threats. Leatherback nesting and marine environments are facing increasing impacts through widespread development and tourism along nesting beaches (Hamann et al. 2006; Hernandez et al. 2007; Maison 2006; Patino-Martinez 2013; Santidrián Tomillo et al. 2007). Structural impacts to beaches include building and piling construction, beach armoring and renourishment, and sand extraction (Bouchard et al. 1998; Lutcavage et al. 1997). In some areas, timber and marine debris accumulation as well as sand mining reduce available nesting habitat (Bourgeois et al. 2009; Chacón Chaverri 1999; Formia et al. 2003; Laurance et al. 2008). Lights on or adjacent to nesting beaches alter nesting adult behavior and is often fatal to emerging hatchlings as they are drawn to light sources and away from the sea (Bourgeois et al. 2009; Cowan et al. 2002; Deem et al. 2007; Witherington 1992; Witherington and Bjorndal 1991). 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). 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. 2009; Gless et al. 2008; Petersen et al. 2009). Wallace et al. (2010) estimated that between 1990 and 2008, at least 85,000 sea turtles were captured as bycatch in fisheries worldwide. This estimate is likely at least two orders of magnitude low, resulting in a likely bycatch of nearly half a million sea turtles annually (Wallace et al. 2010); many of these turtles are expected to be leatherbacks. Donoso and Dutton (2010) found that 284 leatherbacks were bycaught between 2001 and 2005 as part of the Chilean longline fishery, with two individuals observed dead; leatherbacks were the most frequently bycaught sea turtle species. Between 8-17 leatherback turtles likely died annually between 1990 and 2000 in interactions with the California/Oregon drift gillnet fishery; 500 leatherback turtles are estimated to die annually in Chilean and Peruvian fisheries; 200 leatherback turtles are estimated to die in direct harvests in Indonesia; and, before 1992, the North Pacific driftnet fisheries for squid, tuna, and billfish captured an estimated 1,000 leatherback turtles each year, killing about 111 of them each year . Currently, the U.S. tuna and swordfish longline fisheries managed under the HMS FMP are estimated to capture 1,764 leatherbacks (no more than 252 mortalities) for each 3-year period starting in 2007 (NMFS 2004a). 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). Lewison et al. (2004) estimated that 30,000-60,000 leatherbacks were taken in all Atlantic longline fisheries in 2000 (including the U.S. Atlantic tuna and swordfish longline fisheries, as well as others). Use of circle versus traditional J hooks can severely curtail bycatch (Santos et al. 2012) and new regulations are being developed and implemented in several countries around the world for their use.

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).

5.10 Loggerhead sea turtle-South Pacific DPS

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

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

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

Habitat. Small loggerheads, likely hatched on Australian shores, during the austral summer, are found along northern New Zealand (Pritchard 1982a). Most records are from the east coast of the North Island and during the austral winter, with no records during the time of the proposed seismic survey (Gill 1997a; Harley and Kendrick 2006; NZDOC 2014a; NZDOC 2014b).

Reproduction and growth. Loggerhead nesting is confined to lower latitudes temperate and subtropic zones but absent from tropical areas (NMFS and USFWS 1991b; NRC 1990; Witherington et al. 2006b). The life cycle of loggerhead sea turtles can be divided into seven stages: eggs and hatchlings, small juveniles, large juveniles, subadults, novice breeders, first year emigrants, and mature breeders (Crouse et al. 1987). Hatchling loggerheads migrate to the ocean (to which they are drawn by near ultraviolet light Kawamura et al. 2009), where they are generally believed to lead a pelagic existence for as long as 7-12 years (Avens et al. 2013; NMFS 2005). Loggerheads in the Mediterranean, similar to those in the Atlantic, grow at roughly 11.8 cm/yr for the first six months and slow to roughly 3.6 cm/yr at age 2.5-3.5. As adults, individuals may experience a secondary growth pulse associated with shifting into neritic habitats, although growth is generally monotypic (declines with age Casale et al. 2009a; Casale et al. 2009b). Individually-based variables likely have a high impact on individual-to-individual growth rates (Casale et al. 2009b). At 15-38 years, loggerhead sea turtles become sexually mature, although the age at which they reach maturity varies widely among populations (Casale et al. 2009b; Frazer and Ehrhart 1985a; Frazer et al. 1994; NMFS 2001a; Witherington et al. 2006). However, based on new data from tag returns, strandings, and nesting surveys, NMFS (2001a) estimated ages of maturity ranging from 20-38 years and benthic immature stage lasting from 14-32 years.

Loggerhead mating likely occurs along migration routes to nesting beaches, as well as in offshore from nesting beaches several weeks prior to the onset of nesting (Dodd 1988; NMFS and USFWS 1998d). Females usually breed every 2-3 years, but can vary from 1-7 years (Dodd 1988; Richardson et al. 1978). Females lay an average of 4.1 nests per season (Murphy and Hopkins 1984), although recent satellite telemetry from nesting females along southwest Florida support 5.4 nests per female per season, with increasing numbers of eggs per nest during the course of the season (Tucker 2009). Occasional nesting on northern New Zealand shores may occur from May to August (Pritchard 1982a; USFWS 2003).

Migration and movement. Loggerhead hatchlings migrate offshore and become associated with *Sargassum* spp. habitats, driftlines, and other convergence zones (Carr 1986). After 14-32 years of age, they shift to a benthic habitat, where immature individuals forage in the open ocean and coastal areas along continental shelves, bays, lagoons, and estuaries (Bowen et al. 2004; NMFS 2001a). Adult loggerheads make lengthy migrations from nesting beaches to foraging grounds (TEWG 1998). In the Gulf of Mexico, larger females tend to disperse more broadly after nesting than smaller individuals, which tend to stay closer the nesting location (Girard et al. 2009). In the North Atlantic, loggerheads travel north during spring and summer as water temperatures warm and return south in fall and winter, but occur offshore year-round assuming adequate temperature. As water temperatures drop from October to December, most loggerheads emigrate from their summer developmental habitats to warmer waters south of Cape Hatteras, where they winter (Morreale and Standora 1998). For immature individuals, this movement occurs in two patterns: a north-south movement over the continental shelf with migration south of Cape Hatteras in winter and movement north along Virginia for summer foraging, and a not-so-seasonal oceanic dispersal into the Gulf Stream as far north as the 10-15° C isotherm (Mansfield et al. 2009). Wallace et al. (2009) suggested differences in growth rate based upon these foraging strategies. Long Island Sound, Core Sound, Pamlico Sound, Cape Cod Bay, and Chesapeake Bay are the most frequently used juvenile developmental habitats along the Northeast United States Continental Shelf Large Marine Ecosystem (Burke et al. 1991; Epperly et al. 1995a; Epperly et al. 1995b; Epperly et al. 1995c; Mansfield 2006; Prescott 2000; University of Delaware Sea Grant 2000). There is conflicting evidence that immature loggerheads roam the oceans in currents and eddies and mix from different natal origins or distribute on a latitudinal basis that corresponds with their natal beaches (Monzon-Arguello et al. 2009; Wallace et al. 2009). McCarthy et al. (2010) found that movement patterns of loggerhead sea turtles were more convoluted when sea surface temperatures were higher, ocean depths shallower, ocean currents stronger, and chlorophyll a levels lower. Satellite tracking of loggerheads from southeastern U.S. nesting beaches supports three dispersal modes to foraging areas: one northward along the continental shelf to the northeastern U.S., broad movement through the southeastern and mid-Atlantic U.S., and residency in areas near breeding areas (Reina et al. 2012).

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

Gender, age, and survivorship. Although information on males is limited, several studies identified a female bias, although a single study has found a strong male bias to be possible (Dodd 1988; NMFS 2001a; Rees and Margaritoulis 2004). Nest temperature seems to drive sex determination. Along Florida, males primarily derive from earlier-season (LeBlanc et al. 2012). Here, nests ranged from an average sex ratio of 55% female to 85% between 200 and 2004 (LeBlanc et al. 2012). Juvenile and adult age classes have a slight female bias in the central Mediterranean Sea of 51.5% (Casale et al. 2014b).

Additionally, little is known about longevity, although Dodd (1988) estimated the maximum female life span at 47-62 years. Heppell et al. (2003a) estimated annual survivorship to be 0.81 (southeast U.S. adult females), 0.78-0.91 (Australia adult females), 0.68-0.89 (southeast U.S. benthic juveniles, and 0.92 (Australia benthic juveniles). Another estimate suggested a survival rate of 0.41 or 0.60 (CIs 0.20-0.65 and 0.40-0.78, respectively), depending upon assumptions within the study (Sasso et al. 2011). Mediterranean survival is between 0.71 and 0.862 (Casale et al. 2014a). Survival rates for hatchlings during their first year are likely very low (Heppell et al. 2003a; Heppell et al. 2003). Higher fecundity is associated with warmer February and lower May temperatures for loggerheads on the northern Gulf of Mexico (Lamont and Fujisaki 2014).

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

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 1994a; Lenhardt 2002; Moein Bartol and Ketten 2006; Ridgway et al. 1969b). Hearing below 80 Hz is less sensitive but still possible (Lenhardt 1994a). Bartol et al. (1999) reported effective hearing range for juvenile loggerhead turtles is from at least 250-750 Hz. Both yearling and two-year old loggerheads had the lowest hearing threshold at 500 Hz (yearling: about 81 dB re 1 μPa and two-year-olds: about 86 dB re 1 μPa), with thresholds increasing rapidly above and below that frequency (Moein Bartol and Ketten 2006).

These hearing sensitivities are similar to those reported for two terrestrial species: pond and wood turtles. Pond turtles are 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 3000 Hz (Wever and Vernon 1956). Wood turtles are sensitive up to about 500 Hz, followed by a rapid decline above 1000 Hz and almost no responses beyond 3000 or 4000 Hz (Patterson 1966).

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).

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. High temperatures before hatchlings emerge from their nests can also reduce hatchling success, as can bacterial contamination and woody debris in nests (Trocini 2013). Eggs are commonly eaten by raccoons and ghost crabs along the eastern U.S. (Barton and Roth 2008) and in Australia (Trocini 2013). In the water, hatchlings are hunted by herons, gulls, dogfish, and sharks. Heavy loads of barnacles are associated with unhealthy or dead stranded loggerheads (Deem et al. 2009). Brevetoxin-producing algal blooms can result in loggerhead sea turtle death and pathology, with nearly all stranded loggerheads in affected areas showing signs of illness or death resulting from exposure

(Fauquier et al. 2013). The fungal pathogens *Fusarium falciforme* and *F. keratoplasticum* 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).

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 pollution, removal of native vegetation, and planting of non-native vegetation (Baldwin 1992; Margaritoulis et al. 2003; Mazaris et al. 2009b; Patino-Martinez 2013; 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.

The major factors inhibiting their recovery include mortalities caused by fishery interactions and degradation of the beaches on which they nest. Shrimp trawl fisheries account for the highest number of captured and killed loggerhead sea turtles. 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. 2010); many of these are expected to be loggerhead sea turtles.

Marine debris ingestion can be a widespread issue for loggerhead sea turtles. Two thirds of loggerheads contacted marine debris on their way to the ocean and many became severely entangled or entrapped by it (Triessnig et al. 2012). Half of loggerheads in the southwestern Indian Ocean were found to have plastic-based marine debris in their guts (Hoarau et al. 2014).

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). Pike (2014) estimated that loggerhead populations in tropical areas produce about 30% fewer hatchlings than do populations in temperate areas. Historical climactic patterns have been attributed to the decline in loggerhead nesting in Florida, but evidence for this is tenuous (Reina et al. 2013).

Tissues taken from loggerheads sometimes contain very high levels of organochlorines chlorobiphenyl, chlordanes, lindane, endrin, endosulfan, dieldrin, PFOS, PFOA, DDT, and PCB (Alava et al. 2006; Corsolini et al. 2000; Gardner et al. 2003; Guerranti et al. 2013; Keller et al. 2005; Keller et al. 2004a; Keller et al. 2004b; McKenzie et al. 1999; Monagas et al. 2008; Oros et al. 2009; Perugini et al. 2006; Rybitski et al. 1995; Storelli et al. 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). PAH pollution from petroleum origins has been found in Cape Verde loggerheads, where oil and gas extraction is not undertaken in the marine environment (Camacho et al. 2012).

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. 2006a; 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). Elevated mercury levels are associated with deformities in hatchlings versus healthy individuals (Trocini 2013).

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

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

5.11 Olive ridley sea turtle

Populations. Population designations are poorly known. Populations likely correspond somewhat to nesting beach location (Table 10 and Table 11). Most olive ridleys nest synchronously in huge events called "arribadas", with hundreds to thousands of females nesting over the course of three to seven days; other individuals nest alone, out of sequence with the arribada (Aprill 1994b; Kalb and Owens 1994).

Table 10. Estimates of olive ridley turtle arribada size.

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

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

** These data represent total nests for season.

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

Table 11. Locations of olive ridley turtle arribada and solitary nesting beaches in the eastern Pacific and estimates of arribada sizes.

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

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

Southern Hemisphere. Distribution is poorly known, but nesting colonies occur in the Philippines, Papua New Guinea, and northern Australia (Euroturtle 2009; Jensen et al. 2013; Spring 1982). Solitary nesting beaches occur in Australia, Brunei, Malaysia, Indonesia, and Vietnam (Spotila 2004b). Olive ridleys have been sighted in Fiji, Vanuatu, French Polynesia, the Solomon and Marshall islands, and Palau (SPREP 2007). The occurrence of olive ridleys in Tonga and Kiribati is suspected but unconfirmed (SPREP 2007). A population along Australia's Cape York Peninsula appears to be facing extinction (Jensen et al. 2013).

Habitat. Only ten records of olive ridley sea turtles are known for New Zealand waters (NZDOC 2014a; NZDOC 2014b). Occurrence in the area of proposed action area is rare.

Reproduction and growth. Little is known about olive ridley growth or reproduction. However, some beaches, such as Ostional Beach on the Pacific coast of Costa Rica, is known to have extremely low hatching success, particularly at the onset of the dry season onward, at least partly due to the high temperatures of nests (Valverde et al. 2010). Internesting periods of roughly three weeks have been observed in Brazil, with renesting occurring within a few kilometers of the previous nest (Matos et al. 2012). Wind conditions and moon condition appear to be important factors in the timing of mass nesting (Barik et al. 2014). Olive ridley sea turtles along Columbia produce average clutch sizes of 88 eggs, nest twice, incubate for a mean of 65 days, and have average inter-nesting intervals of 18 days (Barrientos-Muñoz et al. 2014). Just over 75% of eggs result in successful newborn emergences (Barrientos-Muñoz et al. 2014).

Migration and movement. Olive ridleys are highly migratory and may spend most of their non-breeding life cycle in deep-ocean waters, but occupy the continental shelf region during the breeding season (Arenas and Hall 1991; Beavers and Cassano 1996; Cornelius and Robinson 1986; Pitman 1991; Pitman 1993; Plotkin 1994; Plotkin et al. 1994a; Plotkin et al. 1995). Reproductively active males and females migrate toward the coast and aggregate at nearshore breeding grounds near nesting beaches (Cornelius 1986; Hughes and Richard 1974; Kalb et al. 1995; Plotkin et al. 1991; Plotkin et al. 1996; Plotkin et al. 1997; Pritchard 1969). Other males and females may not migrate to nearshore breeding aggregations at all (Kopitsky et al. 2000; Pitman 1991). Some males appear to remain in oceanic waters, are non-aggregated, and mate opportunistically as they intercept females *en route* to near shore breeding grounds and nesting beaches (Kopitsky et al. 2000; Plotkin 1994; Plotkin et al. 1994b; Plotkin et al. 1996). Their migratory pathways vary annually (Plotkin 1994), there is no spatial and temporal overlap in migratory pathways among groups or cohorts of turtles (Plotkin et al. 1994a; Plotkin et al. 1995), and no apparent migration corridors exist. Olive ridleys may use water temperature more than any other environmental cue during migrations (Spotila 2004a). Post-nesting migration routes from Costa Rica traverse more than 3,000 km out into the central Pacific (Plotkin et al. 1993; Plotkin et al. 1994a). Olive ridleys from different populations may occupy different oceanic habitats (Polovina et al. 2004; Polovina et al. 2003). Unlike other marine turtles that migrate from a breeding ground to a single feeding area, where they reside until the next breeding season, olive ridleys are nomadic migrants that swim hundreds to thousands of kilometers over vast oceanic areas (Plotkin 1994; Plotkin et al. 1994a; Plotkin et al. 1995). Olive ridleys may associate with flotsam, which could provide food, shelter, and/or orientation cues (Arenas and Hall 1991). In the oceanic eastern tropical Pacific, olive ridley sea turtles are far more common than any other cheloniid (Pitman 1990).

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

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

Status and trends. Except for the Mexico breeding stock, olive ridley sea turtles were listed as threatened under the ESA on July 28, 1978 (43 FR 32800). The olive ridley is the most abundant sea turtle in the world (Pritchard 1997). Worldwide, abundance of nesting female olive ridleys was estimated at two million (Spotila 2004a). Eguchi et al. (2007) counted olive ridleys at sea, leading to an estimate of 1,150,000 – 1,620,000 turtles in the eastern tropical Pacific in 1998-2006

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

Natural threats. Sea turtles face predation primarily by sharks and to a lesser extent by killer whales. Natural predators of olive ridleys also include crabs, garrabos, iguanas, crocodiles, black vultures, wood storks, coyotes, raccoons, and coatis (Aprill 1994a; Bovery and Wyneken 2013). All sea turtles except leatherbacks can undergo “cold stunning” if water temperatures drop below a threshold level, which can pose lethal effects. 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).

Anthropogenic threats. Numerous anthropogenic threats face olive ridley sea turtles (Patino-Martinez 2013). Collection of eggs as well as adult turtles has historically led to species decline (NMFS and USFWS 2007b). Harvests remain a concern for olive ridley recovery. In some locations, takes are now regulated or banned (with varying compliance), while harvests remain uncontrolled in other areas.

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

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

6 ENVIRONMENTAL BASELINE

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 anomolously 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 imminent 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 underice algae to feed on. This may be a cause of decreased krill abundance in the northwestern Antarctic Peninsula during the last decade (Fraser and Hofmann 2003). Meltwaters have also reduced surface water salinities, shifting primary production along the Antarctic Peninsula (Moline et al. 2004). Blue whales, as predators that specialize in eating krill, are likely to change their distribution in response to changes in the distribution of krill (Clapham et al. 1999; Payne et al. 1986; Payne et al. 1990). 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. 2014a). The loss of nesting beaches, by itself, would have catastrophic effects on sea turtle populations globally if they are unable to colonize new beaches that form or if the beaches do not provide the habitat attributes (sand depth, temperature regimes, refuge) necessary for egg survival. In some areas, increases in sea level alone may be sufficient to inundate sea turtle nests and reduce hatching success (Caut et al. 2009a). Storms may also cause direct harm to sea turtles, causing "mass" strandings and mortality (Poloczanska et al. 2009). Increasing temperatures in sea turtle nests alters sex ratios, reduces incubation times (producing smaller hatchling), and reduces nesting success due to exceeded thermal tolerances (Fuentes et al. 2009b; Fuentes et al. 2010; Fuentes et al. 2009c). Smaller individuals likely experience increased predation (Fuentes et al. 2009b).

Climactic shifts also occur due to natural phenomena. In the North Atlantic, this primarily concerns fluctuations in the North Atlantic Oscillation (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).

6.1 Habitat degradation

A number of factors may directly, or indirectly, affect ESA-listed species in the action area by degrading habitat. These include ocean noise and marine debris.

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, construction activities, geophysical surveys, and sonars. In general, ocean background noise levels have doubled every decade for the last six decades, in some areas, primarily due to shipping traffic (IWC 2004). The acoustic noise that commercial traffic contributes to the marine environment is a concern for ESA-listed species because it may impair communication between individuals (Hatch et al. 2008), among other effects (Eriksen and Pakkenberg 2013; Francis and Barber 2013). 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. 2012a).

Marine debris is another significant concern for ESA-listed species and their habitats. Marine debris has been discovered to be accumulating in gyres throughout the oceans. 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).

6.2 Ecotourism

Whale watching is a popular activity in New Zealand waters (Neumann 2001; Orams 2004; Stockin et al. 2008), including the area around Kaikoura where sperm whales are a popular target species for both aerial and vessel-based activities (Lundquist et al. 2012; Richter et al. 2006). These activities are regulated by New Zealand's Marine Mammals Protection Act of 1978 (Richter et al. 2006), as behavioral effects have been noted in New Zealand waters such as vocal patterns, altered diving and jumping behaviors, avoidance, disturbed resting, and changes in swim speed (Lundquist et al. 2012; Orams 2004; Richter et al. 2006; Stockin et al. 2008). Although the action area is near locations of tourism such as Kaikoura, there is not direct overlap.

More generally, individuals may become more vulnerable to vessel strikes once they habituate to vessel traffic (Guzman et al. 2013; Lusseau and Bejder 2007). Several investigators have studied the effects of whale watching vessels on marine mammals (Amaral and Carlson 2005; Christiansen et al. 2013; Corkeron 1995; Felix 2001; Magalhaes et al. 2002; Richter et al. 2003b; Schaffar et al. 2009; Scheidat et al. 2004; Watkins 1986). Whale responses tend to change with vessel characteristics, with responses including:

- Vocalization
- Time at surface
- Swimming speed, swimming angle, or direction
- Respiration rates
- Dive times
- Feeding behavior
- Social interactions

6.3 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 from increasing vessel traffic, and particularly shipping, with increases 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). Although the region does not host extensive commercial vessel traffic, noise emitted from hundreds or thousands of miles away can increase the level of background noise in distant regions, such as the action area. As many as 14 vessels per month pass in or near the action area on average, including cargo and tanker vessels (MarineTraffic 2014; USCG 2013).

Seismic signals emanating from sources a great distance from the action area also contribute to the low frequency ambient sound field (Hildebrand 2009). Baleen whales may be more sensitive to sound at those low frequencies than are toothed whales. Masking of acoustic information can result (Simard et al. 2013); an important issue for marine mammals that rely primarily on sound as a sense. Dunlop et al. (2010) found that humpback whales shifted from using vocal communication (which carries relatively large amounts of information) to surface-active communication (splashes; carry relatively little information) when low-frequency background noise increased due to increased sea state. Other coping mechanisms include shifting the frequency or amplitude of calls, increasing the redundancy or length of calls, or waiting for a quieter period in which to vocalize (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).

6.4 Entrapment and entanglement in fishing gear

Fisheries interactions are a significant problem for several marine mammal species and particularly so for humpback whales. 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 dying 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. Data are not available to address the impact that entrapment or entanglement has on listed whales in the region. However, data from other better-studied regions, such as the North Pacific and Atlantic Oceans, support this being a serious problem globally that we also expect to be present to a degree in the action area. We have no specific information regarding bycatch of sea turtles in New Zealand waters.

6.5 Seismic surveys

At least six seismic surveys have been conducted along the east coast of New Zealand's North Island (Barker et al. 2009; Barnes et al. 2010; Barnes et al. 2011; EOS 2014; Henrys et al. 2013; Pecher et al. 2004). These include:

- 1998-2004 along Hawke's Bay
- 2001 along Hawke's Bay with a 8,238 in^3 array
- 2005 along Hawke's Bay with a 3,840-4,140 in^3 array
- 2007 over the central Hikurangi Margin with a 2,080 in^3 array
- 2011 over the northern Hikurangi Margin
- 2009-2010 in the Pegasus Basin
- 2014 along the eastern North Island using a 6,300 in^3 array
- Among others (NZDOC 2014c; Statoil 2013)

Several of these surveys also included sonar use. An additional seismic survey is pending in the region (LGL 2014).

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

6.6 Ship-strikes

Ship-strike is a significant concern for the recovery of ESA-listed whales. 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. With the information available, we know those interactions occur but we cannot estimate their significance to whale species. This is particularly true for Southern Ocean populations of listed whales, where both fewer people and less traffic likely mean that ship-strike occurs less frequently than in more populated regions, but also means that detecting ship-strike and assessing its impact is also more difficult.

6.7 Scientific and research activities

Scientific research permits issued by the NMFS currently authorize studies of ESA-listed species in the Southern 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. Research activities involve "takes" by harassment, with some resulting mortality. Additional "take" is likely to be authorized in the future as additional permits are issued. It is noteworthy that although the numbers tabulated below represent the maximum number of "takes" authorized in a given year, monitoring and reporting indicate that the actual number of "takes" rarely approach the number authorized. Therefore, it is unlikely that the level of exposure indicated below has or will occur in the near term. However, our analysis assumes that these "takes" will occur since they have been authorized. It is also noteworthy that these "takes" are distributed across the Southern Ocean. Although whales 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. Research "takes" for loggerhead and olive ridley sea turtles have not been authorized in the Southern Ocean. There are numerous permits[6] issued since 2009 under the provisions of both the MMPA and ESA authorizing scientific research on marine mammals and sea turtles. 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.

Tables 8-12 describe the cumulative number of takes for each ESA-listed species in the action area authorized in scientific research permits.

Table 12. Blue whale takes in the Southern Ocean.

Year	Approach	Biopsy	Suction cup tagging	Implantable tagging
2009	609	10	133	10
2010	609	10	133	10
2011	609	10	133	10
2012	389	0	123	0
2013	1,474	150	150	0
2014	1,150	150	150	0
2015	1,150	150	150	0
Total	**5,990**	**480**	**972**	**30**

Permit numbers: 782-1719, 808-1735, 1058-1733, and 14856.

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[6] Permit numbers: 633-1778, 775-1875, 1036-1744, 1058-1733, 10014, 14451, 14856, 15575, 16109, 16239, 16325, 16388, and 17355. See https://apps.nmfs.noaa.gov/index.cfm for additional details.

Table 13. Fin whale takes in the Southern Ocean.

Year	Approach	Biopsy	Suction cup tagging	Implantable tagging
2009	2,799	400	133	10
2010	2,734	400	133	10
2011	2,799	400	133	10
2012	389	0	123	0
2013	1,475	150	150	150
2014	1,150	150	150	150
2015	1,150	150	150	150
Total	**12,496**	**1650**	**972**	**480**

Permit numbers: 782-1719, 808-1735, 1058-1733, and 14856.

Table 14. Humpback whale takes in the Southern Ocean.

Year	Approach	Biopsy	Suction cup tagging	Implantable tagging	Acoustic playback
2009	2,074	140	163	25	0
2010	1,979	150	173	15	0
2011	1,979	50	163	5	0
2012	869	50	163	5	0
2013	1,724	200	160	5	0
2014	2,150	450	160	5	50
2015	2,150	450	160	5	50
Total	**12,925**	**1490**	**1142**	**65**	**100**

Permit numbers: 774-1714, 782-1719, 808-1735, 1058-1733, 14809, 14097, and 14856.

Table 15. Sei whale takes in the Southern Ocean.

Year	Approach	Biopsy	Suction cup tagging	Implantable tagging
2009	1,499	500	123	0
2010	1,499	500	123	0
2011	1,639	500	123	0
2012	389	0	123	0
2013	1,474	150	150	150
2014	1,150	150	150	150
2015	1,150	150	150	150
Total	**8,800**	**1950**	**942**	**450**

Permit numbers: 782-1719, 808-1735, 1058-1733, and 14856.

Table 16. Southern right whale takes.

Year	Approach	Biopsy	Suction cup tagging	Implantable tagging	Acoustic playback
2009	941	120	51	51	0
2010	941	120	51	51	0
2011	460	60	50	50	0
2012	460	60	50	50	0
2013	2810	360	350	300	0
2014	3310	560	350	300	50
2015	3310	560	350	300	50
Total	**15592**	**2400**	**1602**	**1402**	**150**

Permit numbers: 369-1757, 774-1714, 782-1719, 14097, 14809, 14856, and 16239.

Table 17. Sperm whale takes in the Southern Ocean.

Year	Approach	Biopsy	Suction cup tagging	Implantable tagging
2009	5,385	50	10	15
2010	5,285	60	15	15
2011	5,285	75	15	15
2012	225	25	5	5
2013	1,375	175	5	155
2014	1,375	175	5	155
2015	1,375	175	5	155
Total	**20,305**	**735**	**60**	**515**

Permit numbers: 774-1714, 782-1719, 14097, and 14856.

Table 18. Green sea turtle takes in the Southern Ocean.

Year	Capture/handle/ restraint	Sat/sonic/ PIT/flipper tag	Blood/tissue collection
2009	6	6	6
2010	6	6	6
2011	0	0	0
2012	0	0	0
2013	220	220	220
2014	220	220	220
2015	220	220	220
Total	**672**	**672**	**672**

Permit numbers: 1514 and 17022.

Table 19. Hawksbill sea turtle takes in the Southern Ocean.

Year	Capture/handle/ restraint	Sat/sonic/ PIT/flipper tag	Blood/tissue collection
2009	0	0	0
2010	0	0	0
2011	0	0	0
2012	0	0	0
2013	165	165	165
2014	165	165	165
2015	165	165	165
Total	**495**	**495**	**495**

Permit numbers: 17022.

Table 20. Leatherback sea turtle takes in the Southern Ocean.

Year	Capture/handle/ restraint	Sat/sonic/ PIT/flipper tag	Blood/tissue collection
2009	1	1	1
2010	1	1	1
2011	1	1	1
2012	1	1	1
2013	1	1	1
2014	1	1	1
2015	1	1	1
Total	**7**	**7**	**7**

Permit numbers: 14381.

6.8 Impacts of the Environmental Baseline on 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 area. Federal projects in the action area that have already undergone formal or early section 7 consultation, and state or private actions that are contemporaneous with this consultation, 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 on the information available to us in this section.

Climate change has wide-ranging impacts, some of which can be experienced by ESA-listed whales and sea turtles 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. Changing sea levels will inundate sea turtle nesting beaches. Considering the sensitivity that baleen whales and some sea turtles have to ephemeral, varying oceanographic patterns, warming water temperatures change the location and timing of foraging grounds, the effort required to find sufficient forage, and the migration patterns whales and sea turtles undertake to and from these areas.

Acoustic effects from anthropogenic sources, whether they are vessel noise or seismic sound could also have biologically significant impacts to ESA-listed whales and sea turtles 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 is a small fraction of the total exposure received, it is expected to occur in rare instances.

Several factors that are likely important morbidity and mortality factors for ESA-listed whales and sea turtles (entrapment, entanglement, ship-strike, and marine debris ingestion) are poorly or not known for the action area and the Southern Ocean in general. However, we do expect that fishing gear, marine debris, and vessel traffic occur in the action area and the surrounding region that can result in injury and potentially death of individuals as they do in better-studied areas, such as the North Pacific and Atlantic Oceans.

Authorized research and tourist-related activities 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 two to four kilometers 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. 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.

7 EFFECTS OF THE PROPOSED ACTIONS

Pursuant to section 7(a)(2) of the ESA, Federal agencies must insure, in consultation with NMFS, that their activities are not likely to jeopardize the continued existence of any ESA-listed species or result in the destruction or adverse modification of critical habitat. The proposed NSF and SIO support of the seismic survey and issuance of the incidental harassment authorization for “takes” of marine mammals during the seismic study would expose ESA-listed species to seismic airgun pulses, as well as sound emitted from a multi-beam bathymetric echosounder and sub-bottom profiler as well as other stressors. In this section, we describe the potential physical, chemical, and biological stressors associated with the proposed actions, the probability of individuals of ESA-listed species being exposed to these stressors, and the probable responses of those individuals (given probable exposures) based on the best available scientific and commercial evidence. As described in the *approach to the assessment* section, for any responses that are expected to reduce an individual’s fitness (i.e., growth, survival, annual reproductive success, or lifetime reproductive success), the assessment would then consider the risk posed to the viability of the population(s) those individuals comprise and to the ESA-listed species those populations represent. The purpose of this assessment and, ultimately, of the Opinion is to determine if it is reasonable to expect the proposed action to have effects on ESA-listed species that would be reasonably expected to appreciably reduce their likelihood of surviving and recovering in the wild.

For this consultation, we are particularly concerned about behavioral and physiological disruptions that may result in animals that fail to feed or breed successfully or fail to complete their life cycle because these responses are likely to have population-level consequences. The proposed IHA action would authorize non-lethal “takes” by harassment as defined by the MMPA of ESA-listed species during seismic survey activities. The ESA does not define harassment, nor has NMFS done so through regulation. The MMPA defines harassment as any act of pursuit, torment, or annoyance that 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 US Fish and Wildlife Service’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

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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)

patterns that are essential to the animal's life cycle or its contribution to the population the animal represents.

Our analysis considers that behavioral harassment or disturbance is not limited to the 160 dB acoustic "take" definition for marine mammals (non-continuous and continuous noise exposure levels, respectively) and may, in fact, occur in many ways. Fundamentally, if our analysis leads us to conclude that an individual changes its behavioral state (for example, from resting to traveling away from the airgun source or from traveling to evading), we consider the individual to have been harassed or disturbed, regardless of whether it has been exposed to acoustic sources at levels that define "take," as long as it creates the probability of injury. In addition, individuals may respond in a variety of ways, some of which have more significant fitness consequences than others. For example, quick evasion of a seismic source would be more significant than slow travel away from the same stressor, due to increased metabolic demands, stress responses, and potential for calf abandonment that this response could or would entail. As described in the *approach to the assessment*, the universe of likely responses is considered in evaluating whether those responses lead to fitness consequences for the individual and (if appropriate), the affected population and species as a whole to determine the likelihood of jeopardy.

7.1 Potential Stressors

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

1. Pollution by oil or fuel leakage
2. Acoustic interference from engine noise
3. Ship-strikes
4. Entanglement in towed hydrophone cable and bottom sampling equipment (e.g., piston core/gravity core, heat-flow probe)
5. Sound fields produced by airguns
6. Sound fields from sub-bottom profiler and multibeam echosounder

7.1.1 Pollution by Oil or Fuel Leakage

Based on a review of available information, we determined which of these possible stressors would be likely to occur and which would be discountable or insignificant.

The potential for biologically-meaningful fuel or oil leakages is extremely 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 *Revelle* 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.

7.1.2 **Disturbance from Engine Noise**

Vessel noise has the potential to affect ESA-listed species. Although noise originating from vessel propulsion will propagate into the marine environment, this amount would be small. The *Revelle* will be traveling at generally slow speeds (9 km/h), reducing the amount of noise produced by the propulsion system. The *Revelle* will frequently stop or move short distances at relatively slow speeds in association with oceanographic equipment deployment, operation, and retrieval. The *Revelle*'s passage past a whale would be brief and not likely to be significant in affecting 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.

7.1.3 **Ship Strike**

The *Revelle* 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 *Revelle* has traveled without a ship strike, general expected movement of marine mammals away or parallel to the *Revelle*, as well as the generally slow movement of the *Revelle* during most of its travels. All things considered, we have concluded the potential for ship strike or acoustic interference from propulsion and machinery noise 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.

7.1.4 Entanglement

ESA-listed species could interact directly with the towed hydrophone streamers and deployed oceanographic equipment and these interactions have been documented. However, such events with ESA-listed marine mammals are unknown to us. Although the towed hydrophone streamers or oceanographic equipment lowered in a controlled manner over-the-side could come in direct contact with an ESA-listed species, entanglements are highly unlikely and considered improbable based on investigation into the use of these devices during the activities of other oceanographic activities. Given this, we expect that the risk of entanglement in towed hydrophone cable or other oceanographic equipment so low as to be discountable. Therefore, it is not likely to adversely affected ESA-listed species and will not be considered further in this Opinion.

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, ADCP, and multibeam echosounder sonars.

7.1.5 Stressors Considered Further in this Opinion

This consultation focused on the following stressors produced by the proposed seismic activities that are likely to 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.

7.2 Exposure Analysis

Exposure analyses identify the physical, chemical, and biotic stressors produced by a proposed action that could 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 fields 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.

The Permits and Conservation Division applies acoustic thresholds to help determine at what point during exposure to seismic airguns (and other acoustic sources) marine mammals are considered "harassed" under the MMPA. For this consultation, we adopted the same thresholds to estimate the number of exposures of ESA-listed marine mammals (i.e., blue, sei, fin, humpback, southern right, and sperm whales) to seismic airguns at a level that would be harassment under the ESA. These thresholds also were used to develop exclusion radii around the source and the necessary power-down or shut-down criteria to limit marine mammals and sea turtles' exposure to harmful levels of sound.

The NSF provided an estimate of the number of marine mammals that would be exposed to the levels of sound in which they would be considered "taken" during the proposed survey. Additionally, the Permits and Conservation Division conducted an independent exposure analysis that was informed by comments received during the public comment period on the draft environmental assessment and the proposed IHA notice. In this section we describe both of those 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.

The NSF and NMFS's Permits and Conservation Division estimated the number of ESA-listed whales exposed to received levels ≥160 dB re: 1 µPa$_{rms}$ for non-continuous sound (airguns and sonars). The chosen method calculated the product of animal density and ensonified area, and was developed in cooperation with NMFS Permits and Conservation Division. We agree that this is the best available approach and adopt it in developing the anticipated reasonable level of exposure within this Opinion. Maximum radii associated with seismic airgun isopleth modeling were established at the maximum diving depth for ESA-listed species (2,000 m). Because of the biological information presented in the *Status of Listed Resources* indicating that most ESA-listed whales do not dive to this depth and, for those that do, we expect that individuals will rarely be found at this depth, the isopleth distance from the source array is likely to overestimate the exposure ESA-listed individuals are expected to experience. However, we feel this conservative assumption is appropriate to carry forward, given uncertainty associated with elements not included in the isopleth modeling (such as bottom topography, sound channel characteristics, bathymetric characteristics, water chemistry, and other propagation features). A recent study supports this, showing that in shallow water, measured power levels and signal length can vary based on bathymetric features (Crone 2014).

Although the action area includes the region ensonified by airguns to the point at which the anthropogenic sound decreases to ambient levels, we expect a subset of this area to have effects that are more significant. We expect responses to seismic sound sources by ESA-listed marine mammals occur within the 160 dB re: 1 µPa$_{rms}$ (modeled to be up to 600 m from the *Revelle*'s two-airgun array along 1,563 km of trackline) isopleth. We expect the ensonified area to at least 160 dB re: 1 µPa$_{rms}$ along the trackline to roughly 1,153.6 km^2 for airguns, and to at least 166 dB re: 1 µPa$_{rms}$ along the trackline to about 750 km^2.

7.2.1 Marine Mammal Exposure

The NSF and SIO proposed a variety of data sources to base marine mammal density estimates on. The NSF, SIO, NMFS's Permits and Conservation Division, and ESA Interagency Cooperation Division ultimately agreed that the superior data set to use for all ESA-listed whales (excluding southern right whales) was based on the IWC/IDCR survey data (Butterworth et al. 1994) for Regions V and IV. We excluded species density estimates for the Southern Ocean that were extrapolated from Japanese scouting vessels (see discussion below).The NSF and SIO did not provide exposure estimates for ESA-listed sea turtles. To obtain estimates of exposure for sea turtles, we relied upon summarized records of marine reptiles in New Zealand from 1837 to 1996 (Gill 1997a) and reports from the New Zealand Department of Conservation Atlas (NZDOC 2014a; NZDOC 2014b).

The NSF and SIO estimated the exposure radii around the proposed *Revelle* operations using empirical data gathered in the Gulf of Mexico in 2007-2008 aboard the R/V *Langseth* and modeling based on these data. The maximum distances from airguns where received levels might reach 160 dB re: 1 µPa$_{rms}$ for a two-airgun array at 2,000 m depth (maximum depth at which

ESA-listed species are expected to occur) in deep water at 3 m tow depth are summarized in Table 1 on page 7. A thorough review of available literature (see *Response Analysis*) supports these as average received levels baleen whales tend to show some avoidance response to received seismic sound.

The NSF's and SIO's assumption that individuals will move away if they experience sound levels high enough to cause significant stress or functional impairment is also reasonable (see *Response Analysis*). Isopleth modeling tends to overestimate the distance to which various isopleths will propagate and expose ESA-listed individuals because most exposure will likely occur at depths shallower than 2,000 m, where received sound levels should be reduced (see Figure 2). Because we are unable to know where individuals will be in the water column at the time of exposure, we accept this assumption.

Through consultation, the NSF, SIO, NMFS' Permits and Conservation Division, and ESA Interagency Cooperation Division agreed that the fore-mentioned approach was the best available and worked to identify the necessary information for this analysis.

Exposure of Listed Mammals to Airguns. NSF exposure estimates (

Table 21) were calculated by using the density per 1,000 km^2. These densities were multiplied by the ensonified area (1,153.6 km^2 ensonified to the 160 dB re: 1 μPa_{rms} level) to obtain the total number of exposures (rounded to the next whole number). Based on the quality of the data, the ESA Interagency Cooperation Division and the Permits and Conservation Division believe that the use of Butterworth et al. (1994) IWC Southern Ocean Whale and Ecosystem Research program (blue, fin, sei, humpback, and sperm whales) density data is the best available data to use here. Therefore, we used the same density estimates as the Permits and Conservation Division in determining exposure estimates greater than 160 dB re: 1 μPa_{rms}. After discussion with the Permits and Conservation Division, sperm whale estimates were increased to mean group size (Childerhouse et al. 1995).

The NSF, SIO, NMFS's Permits and Conservation Division, and ESA Interagency Cooperation Division ultimately agreed that the superior data set to use for all ESA-listed whales (excluding southern right whales) was based on the IWC/IDCR survey data (Butterworth et al. 1994) for Regions V and IV. We excluded species density estimates for the Southern Ocean that were extrapolated from Japanese scouting vessels. We decided to exclude these data because the Japanese scouting data were collected in a different time of year (January and February) and on a latitudinal basis, all across the globe, with no way to get at more specific regions closer to the action area. The IWC/IDCR data were collected in a timeframe and in areas where that were more applicable to the proposed action.

Because these data did not include southern right whales, the NSF adopted a secondary data source based on survey data from the U.S. west coast (Barlow 2010; Forney 2007); the Permits and Conservation Division adopted this source as well. Although density estimates are not quantitatively available for New Zealand waters, we expect southern right whales to be present in the action area and exposed to the seismic activities, as the species is known to winter in New Zealand (Patenaude 2000; Patenaude 2003a) (see *status of listed resources* section).

We relied upon studies within the action area to estimate southern right whale exposure. Southern right whales are known to spend austral winters in coastal New Zealand waters, with females giving birth to and nursing their young; a recent study calculated a mean group size of 1.7 (Rayment et al. 2012). We rounded this up to the next whole number to obtain an exposure estimate of two southern right whales.

Table 21. Estimated exposure of ESA-listed whales to sound levels ≥160 dB re: 1 μPa$_{rms}$ during the proposed seismic activities developed from International Whaling Commission Southern Ocean Whale and Ecosystem Research program density data.

Whale density per 1,000 km^2	# of exposures to listed whales	# of whales exposed to activities	Population size	% of population exposed	Population/ location
Blue 0.046648	2	Up to 2	1,700[1]	Up to 0.12%	Southern Hemisphere
Fin 0.1329	2	Up to 2	15,178[2]	Up to 0.02%	South of 30°S
Sei 0.0781354	2	Up to 2	9,718[2]	Up to 0.03%	South of 60°S
Humpback 0.2466513	2	Up to 2	3,500[2]	Up to 0.06%	2,300-3,500/Oceania
Sperm 1.1577	10	Up to 10	128,000[3]	Up to 0.01%	South of 30°S
Southern right	2	Up to 2	2,619[4]	Up to 0.08%	New Zealand
Total		--	--	--	--

IWC 2005c

[1] *IWC (1996)*

[2] (Constantine et al. 2010)

[3] *(Butterworth et al. 1995) as cited in (Perry et al. 1999)*

[4] Carroll et al. 2013

Whales of all age classes are likely to be exposed. Based on our understanding of ESA-listed whale life history presented in the *status of listed resources*, ESA-listed whales are expected to be feeding, traveling, or migrating in the area, and some females would have young-of-the-year accompanying them. We normally assume that sex distribution is even in whales and sexes are exposed at a relatively equal level. However, sperm whales in the area likely consist of lone or small male groups than groups of females. Therefore, we expect a male bias to sperm whale exposure.

Exposure of ESA-listed whales to multibeam echosounder and sub-bottom profiler. Two additional acoustic systems will operate during the proposed *Revelle* 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, 3.5 kHz for the sub-bottom profiler). As such, their frequencies will attenuate more rapidly than those from airgun sources will. Listed individuals would experience higher levels of airgun noise well before multibeam echosounder or sub-bottom profiler noise of equal amplitude would reach them. When airguns are not operational, sonars would still be active. For sonars that are audible, the slow movement of the *Revelle* and continuous operation of the sonars would alert ESA-listed whales to the vessel's presence and, if the *Revelle* approaches more closely, continually serve as a notice of the vessel's movement. As with airguns, if received sound levels begin to reach levels that are physiologically challenging, we expect a stress response may be initiated and for animals to move away.

While airguns are not operational, marine mammal observers will remain on duty to collect sighting data. If ESA-listed whales were to closely-approach the vessel, the *Revelle* would take evasive actions to avoid a ship-strike as well as lessen exposure to very high source levels. We 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 used during the proposed activities presented a low risk for auditory damage or any other injury, and that an individual would require exposure to 250–1,000 pulses from a sub-bottom profiler to be at risk for a temporary threshold shift (TTS). To be susceptible to TTS, a whale would have to pass at very close range and match the vessel's speed; we expect a very small probability of this during the proposed study. An individual would have to be well within 100 m of the vessel to experience a single multibeam echosounder pulse that could result in TTS (LGL Ltd. 2008). The same result could only occur at even closer ranges for sub-bottom profiler signals, because the signals are weaker. Furthermore, we expect both multibeam echosounder and sub-bottom profiler systems to operate continuously. It is possible, however, that some small number of listed whales (fewer than those exposed to airguns) could experience low-level multibeam echosounder and/or sub-bottom profiler sound. We are unable to quantify the level of exposure, but do not expect any exposure to result to occur at high levels.

7.2.2 Mitigation Measures

A major mitigation factor proposed by the NSF is visual monitoring combined with power- and shut-down, especially for marine mammals, which should reduce exposure of listed whales and sea turtles. However, visual monitoring has several limitations. Although areas ensonified by 160, 166, and 180 dB re 1 μPa$_{rms}$ propagation distances are within the visual range of the *Revelle* and its observers, it is unlikely that all listed species are at the surface and visible at these distances. Vessel platforms are subject to some limitations such as that even under good sighting conditions, observers have limited ability to identify protected species during their brief time at the surface.

On their own, power-down and shut-down procedures are unlikely to be completely effective at eliminating the co-occurrence of listed individuals within the sound field ≥160 dB re 1 μPa$_{rms}$. Other measures such as vessel turns and minimizing airgun source levels, seek to further minimize the exposure ESA-listed species will experience. Ramp-up was effective in reducing hearing-related effects in sonar systems (Von Benda-Beckmann et al. 2014) and we expect reduced or less intense exposure with application of airgun ramp-up.

When combined with the other proposed mitigation and monitoring measures, we conclude that the probability of ESA-listed individuals being exposed to the sound field ≥ 160dB re 1 μPa$_{rms}$ is reduced by the use of ramp-ups and shut-downs. Vocalizations by protected species will also help in identifying abundance of cetaceans in the action area. Further ability to identify bearing, distance, and abundance is limited.

7.2.3 Re-exposure

For all ESA-listed species, the NSF, SIO, and NMFS' Permits and Conservation Division did not provide a rationale in their environmental assessments for their levels of re-exposure to the same individual. The ESA Interagency Cooperation Division expects each exposure would generally be to a unique animal, rather than re-exposure of the same animal, multiple times. This rationale is based on, among other things, little overlap in one trackline's ensonified area with another. The structures of the tracklines influence how much overlap there is between the ensonified areas; for this survey, the tracklines are widely-spaced, meaning that there is very little overlap between the ensonified areas.

It is reasonable to expect, based on review of observed effects of seismic sound exposure to marine mammals, that some individuals will move a distance of several hundred meters to tens of kilometers away, due to individual or situational sensitivity, or other rationale for why whales move (ex. feeding, migration, or breeding opportunities unrelated to effects of the proposed action). As such, it is reasonable to expect that some individuals will receive a single exposure and vacate the action area by moving away from the immediate area of the sound field. Other individuals may move, but to locations where re-exposure could occur, either due to the direction or short distance they travel. Observations from previous seismic surveys support the likelihood that individuals will be re-exposed rarely, if at all. We also expect at least some individuals will return to the area after the seismic activity has ceased. We expect the only occasions when re-exposure may occur is when individuals move away and happen to place themselves on another portion of the seismic survey trackline, or in the infrequent instances when tracklines cross. This is particularly significant given that marine mammals tend to return to specific areas for foraging and breeding, or use particular migratory corridors. However, based on observations from previous seismic surveys and our professional judgment, the likelihood that individuals will be re-exposed several times is low. We consider this is unlikely to happen in other than random, rare cases and we expect the vast majority of individuals would only be exposed once. We also considered that ESA-listed whales will likely be migrating or traveling through the region and not consistently occurring in the same place. For those that do remain in place (potentially individuals socializing or pursuing foraging opportunities), these individuals will also, in most cases, be moving with relatively constant and rapid current features. The *Revelle*, however, will use GPS technology to follow the exact path in reshooting lines. This means that animals would through drift, actively, or passively move from their previous location and not be re-exposed in the same way they were initially. However, given that some locations within the region may be ensonified to levels that may cause biologically-meaningful responses (160 dB re: 1 μPa_{rms} or higher) up to two times, we expect that a single individual may be exposed up to two times to this level or higher.

7.2.4 Sea turtles Exposure

Exposure of listed turtles to airguns. 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 occurrence and a planned ensonified area of approximately 750 km^2 along survey track lines, including areas of repeated exposure from adjacent track lines and turning legs. 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”.

NSF presented estimated distances for the 166 dB re 1 μPa_{rms} sound levels generated by the two 45 in^3 GI-guns. In deep water (>1000 m), the predicted established distance at received levels is 200 m; in shallower water (100-1000 m), this distance is 300 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 five ESA-listed sea turtle species that are likely to be affected by the proposed action: green, hawksbill, olive ridley, leatherback, and loggerhead South Pacific 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 New Zealand waters. To estimate exposure for sea turtles, we relied on a paper which summarized records of marine reptiles in New Zealand from 1837 to 1996 (Gill 1997a) and reports from the New Zealand Department of Conservation Atlas (NZDOC 2014a; NZDOC 2014b).

Leatherbacks and green sea turtles are most common in New Zealand waters (NZDOC 2014b), so we expect those species to be the most likely to be exposed. Hawksbill and loggerhead sea turtles are present in the action area, but to a lesser degree. Olive ridley sea turtles are considered rare[8], so we expect a lower likelihood of exposure for this species compared to the other turtles.

The proposed seismic survey will take place between May and June. Gill (1997a) reported the distribution by month of records for each sea turtle species (Table 22). There were seven records of leatherbacks reported in May and June, four records for hawksbill, and three for green sea turtles (Gill 1997a). There were no loggerhead sea turtle records reported for May and June, although records exist for other months of the year. The maximum number of loggerhead reports from a single month occurred in July (n=4). Olive ridley sea turtles are considered rare in New Zealand, with only four live turtles being sighted throughout the country since 1988 (NZDOC 2014a; NZDOC 2014b).

Table 22. Number of ESA-listed sea turtles sighting records in New Zealand waters, taken from Gill 1997a (1837-1996) and the NZ DOC Atlas (1988-present).

Sea turtle species	# of sea turtles sighted in May and June (Gill 1997a)	# of sea turtles sighted alive and dead, 1988-present† (NZ DOC Atlas)
Leatherback	7	62/25
Hawksbill	4	18/5
Green	3	45/17
Loggerhead	0	18/10
Olive Ridley	0	4/5

There are limitations to using this information to quantify exposure for the proposed action. First, the records from Gill (1997a) are not sorted by location, so it is possible that these records come from areas outside the proposed action area. In fact, the information from the NZ DOC Atlas indicates that many sea turtle sightings occur on the northern and western shores of the North Island (i.e., hundreds of kilometers away from the proposed action). However, since the information in Gill (1997a) is specific to a month (if not a location), this at least allows us to develop a reasonable probability of when we expect these sea turtles to be present in the action

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[8] http://www.doc.govt.nz/nature/native-animals/marine-fish-and-reptiles/sea-turtles/

†Pre-1988 records are also available in the NZ DOC atlas NZDOC. 2014a. Electronic atlas of the amphibians & reptiles of New Zealand, NZDOC. 2014b. Sea turtles in New Zealand.. They were omitted here because those records would likely be captured in Gill 1997a summary.

area. Secondly, the Gill (1997a) reports occur over a very broad timespan (1837-1996), and combine sea turtle sightings from across all years; the New Zealand DOC Atlas reports also combine sea turtle sightings across years (1988 to present and pre-1988). The seismic activities will occur over 30 days in May and June. Therefore, we believe it would be inappropriate to use the combined sea turtle sightings to represent exposure numbers, as that would not be an accurate reflection of what we expect to occur in the comparatively small action area and over a shorter time period than what is presented by Gill 1997a and in the NZ DOC Atlas reports.

No known nesting occurs within the action area, so we do not expect hatchlings to be exposed. The waters of the action area during austral fall and winter are also relatively cold for sea turtles and likely preclude the ability of young sea turtle age classes to survive here. We expect that adult non-leatherbacks or subadult to adult leatherback sea turtle age classes may be present and exposed. Although sufficient information to address differences in sex ratio in the action area are lacking, we expect that the general findings from other locations likely hold in the action area; leatherback exposure would be heavily skewed towards females (Binckley et al. 1998; James et al. 2007; Plotkin 1995), loggerheads moderately skewed towards males (Dodd 1988; NMFS 2001a; Rees and Margaritoulis 2004), and green sea turtles skewed towards females (Wibbels 2003).

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

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

7.3 Response Analysis

As discussed in the *Approach to the assessment* section of this Opinion, response analyses determine how listed resources are likely to respond after exposure to a stressor created by the action in the action area. Our response analysis attempts to detect potential lethal, sub-lethal (or physiological), or behavioral responses that might result in reducing the fitness of listed individuals. Ideally, response analyses would consider and weigh evidence of adverse consequences, as well as evidence suggesting the absence of such consequences.

7.3.1 Potential Response of Marine Mammals to Acoustic Sources

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

- Hearing threshold shifts
- Auditory interference (masking)
- Behavioral responses
- Non-auditory physical or physiological effects

The *Response Analysis* also considers information on the potential for stranding and the potential effects on the prey of ESA-listed whales in the action area.

Marine mammals and threshold shifts. Exposure of marine mammals to very strong sound pulses can result in physical effects, such as changes to sensory hairs in the auditory system, which may temporarily or permanently impair hearing. 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 from 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 shifts (PTSs) 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, 2005 #61997}{Schlundt, 2000 #62578}{Ketten, 2012 #94892}.

Few data are available to precisely define each ESA-listed species' hearing range, let alone its sensitivity and levels necessary to induce TTS or PTS. Based on 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. 2007b). Additional studies have since been conducted to further refine this (Nachtigall and Supin 2013; Popov et al. 2013a; Popov et al. 2013b; Schlundt et al. 2013; Supin et al. 2013; Tougaard et al. 2013). PTS is expected at levels ~6 dB greater than TTS levels on a peak-pressure basis, or 15 dB greater on an SEL basis than TTS (Southall et al. 2007b). In terms of exposure to the *Revelle*'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 36 m of the *Revelle*'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 μPa^2s.

For several reasons, we do not expect TTS or PTS to occur to any ESA-listed whale because of airgun exposure. We expect that individuals will move away from the airgun array as it approaches. We further believe that as sound intensity increases, individuals will experience conditions (stress, loss of prey, discomfort, etc.) that prompt them to move away from the sound source, and thus avoid exposures that would induce TTS. Ramp-ups would also reduce the probability of TTS-inducing exposure at the start of seismic surveys. Furthermore, mitigation measures would be in place to initiate a power-down if individuals enter, or are about to enter, the 180 dB isopleth or within 111 m during airgun operations, which is below the levels believed to be necessary for potential TTS.

Marine mammals and auditory interference (masking). Interference, or masking, generally occurs when the interfering noise is of a similar frequency and similar to or louder than the auditory signal received by an animal processing echolocation signals or listening for acoustic information from other individuals (Francis and Barber 2013). Masking can interfere with an individual's ability to gather acoustic information about its environment, such as predators, prey, conspecifics, and other environmental cues. Generally, noise will only mask a signal if it is sufficiently close to the signal in frequency. This can result in loss of environmental cues of predatory risk, mating opportunity, or foraging options (Francis and Barber 2013). Low frequency sounds are broad and tend to have relatively constant bandwidth, whereas higher frequency bandwidths are narrower (NMFS 2006h).

There is frequency overlap between airgun noise and vocalizations of ESA-listed whales, particularly baleen whales but also sperm whales. Any masking that might occur would likely be temporary because seismic sources are not continuous and the seismic vessel would continue to transit. The proposed seismic surveys could mask whale calls at some of the lower frequencies. 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 *Revelle*'s airguns). Given the disparity between sperm whale echolocation and communication-related sounds with the dominant frequencies for seismic surveys, masking is not likely to be significant for sperm whales (NMFS 2006h). Overlap of the dominant low frequencies of airgun pulses with low-frequency baleen whale calls are expected to pose a somewhat greater risk masking. The *Revelle's* airguns will emit a 0.1 sec pulse when fired every 5-10 sec. Therefore, pulses will not "cover up" the vocalizations of listed whales to a significant extent (Madsen et al. 2002). Individuals can continue calling and be heard between airgun pulses (Nieukirk et al. 2012). We address the response of listed whales stopping vocalizations because 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 (Gedamke and McCauley 2011; Guerra et al. 2013). 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 (Gedamke and McCauley 2011; Guerra et al. 2013; Guerra et al. 2011), potentially interfering with the ability of animals to hear otherwise detectible sounds in their environment. Wittekind et al. (2013) estimated that blue and fin whales may have their communication range reduced by 2,000 km.

Marine mammals and behavioral responses. We expect the greatest response to airgun sounds by number of responses and overall impact to be in the form of changes in behavior. Listed individuals may briefly respond to underwater sound by slightly changing their behavior or relocating a short distance, so the effects can equate to a 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 ESA-listed whales considered in this Opinion directly, additional studies to other related whales (such as bowhead and gray whales) are relevant in determining the responses expected by species under consideration. Therefore, studies from non-listed or species outside the action area are also considered here. Individual differences in responding to stressful stimuli also appear to exist, and appear to have at least a partial genetic basis in trout (Laursen et al. 2011). Animals generally respond to anthropogenic perturbations as they would predators, increasing vigilance and altering habitat selection (Reep et al. 2011). Habitat abandonment due to anthropogenic noise exposure has been found in terrestrial species (Francis and Barber 2013).

Several studies have aided in assessing the various levels at which whales may modify or stop their calls in response to airgun sound. Whales continue calling while seismic surveys are operating locally (Greene Jr et al. 1999; Jochens et al. 2006; Madsen et al. 2002; McDonald et al. 1993; McDonald et al. 1995a; Nieukirk et al. 2004; Richardson et al. 1986; Smultea et al. 2004; Tyack et al. 2003). However, humpback whale males increasingly stopped vocal displays on

Angolan breeding grounds as received seismic airgun levels increased (Cerchio et al. 2014). Some blue, fin, and sperm whales stopped calling for short and long periods apparently in response to airguns (Bowles et al. 1994; Clark and Gagnon 2006; McDonald et al. 1995a). Fin whales (presumably adult males) engaged in singing in the Mediterranean Sea moved out of the area of a seismic survey while airguns were operational as well as for at least a week thereafter (Castellote et al. 2012). A blue whale discontinued calls in response to received airgun sound of 143 dB re: 1 μPa for one hour before resuming (McDonald et al. 1995a). Blue whales may also attempt to compensate for elevated ambient sound by calling more frequently during seismic surveys (Iorio and Clark 2009). Sperm whales, at least under some conditions, may be particularly sensitive to airgun sounds, as they have been documented to cease calling in association with airguns being fired hundreds of kilometers away (Bowles et al. 1994). Other studies have found no response by sperm whales to received airgun sound levels up to 146 dB re: 1 $\mu Pa_{p\text{-}p}$ (Madsen et al. 2002; McCall Howard 1999). Other individuals may increase vocalization source levels or shift the frequencies those vocalizations occur at (Tyack and Janik 2013). Some exposed individuals may cease calling in response to the *Revelle*'s airguns. If individuals ceased calling in response to the *Revelle*'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. 1999b). Surface duration decreased markedly during seismic sound exposure, especially while individuals were engaged in traveling or non-calf social interactions (Robertson et al. 2013). Migrating bowhead whales show strong avoidance reactions to received 120–130 dB re: 1 μPa_{rms} exposures at distances of 20-30 km, but only changed dive and respiratory patterns while feeding and showed avoidance at higher received sound levels (152–178 dB re: 1 μPa_{rms}) (Harris et al. 2007; Ljungblad et al. 1988; Miller et al. 1999; Miller et al. 2005; Richardson et al. 1995c; Richardson et al. 1999b; Richardson et al. 1986). Responses such as stress may occur and the threshold for displacement may simply be higher while feeding. Bowhead calling rate was found to decrease during migration in the Beaufort Sea as well as temporary displacement from seismic sources (Nations et al. 2009). Bowheads were found to be less sightable during airgun exposure than at other times due to altered dive patterns (Robertson 2014). Calling rates decreased when exposed to seismic airguns at received levels of 116-129 dB re: 1 μPa (possibly but not knowingly due to whale movement away from the airguns), but did not change at received levels of 99-108 dB re: 1 μPa (Blackwell et al. 2013). Despite the above information and exposure to repeated seismic surveys, bowheads continue to return to summer feeding areas and when displaced, bowheads appear to reoccupy areas within a day (Richardson et al. 1986). We do not know whether the individuals exposed in these ensonified areas are the same returning or whether individuals that tolerate repeat exposures may still experience a stress response.

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

Humpback whales 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 8-12 km from the seismic source (McCauley et al. 2000a; McCauley et al. 1998). A startle response occurred as low as 112 dB re: 1 μPa_{rms}. Closest approaches were generally limited to 3-4 km, although some individuals (mainly males) approached to within 100 m on occasion where sound levels were 179 dB re: 1 μPa_{rms}. Changes in course and speed generally occurred at estimated received level of 157–164 dB re: 1 μPa_{rms}. Feeding humpbacks appear to be somewhat more tolerant. Humpback whales along Alaska startled at 150–169 dB re: 1 μPa and no clear evidence of avoidance was apparent at received levels up to 172 re: 1 μPa_{rms} (Malme et al. 1984; Malme et al. 1985). Potter et al. (2007) found that humpbacks on feeding grounds in the Atlantic did exhibit localized avoidance to airguns. Among humpback whales on Angolan breeding grounds, no clear difference was observed in encounter rate or point of closest approach during seismic versus non-seismic periods (Weir 2008).

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 detect 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).

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). This has been a general observation of large whales (excluding sperm whales) for several seismic surveys off eastern Canada (Moulton and Hols 2010). 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 ESA-listed whales exposed to seismic airgun sound will exhibit an avoidance reaction, displacing individuals from the area at least temporarily. We also expect secondary foraging areas to be available that would allow whales to continue feeding. Although breeding may be occurring, we are unaware of any habitat features that whales would be displaced from that are essential for foraging if whales depart an area as a consequence of the *Revelle*'s presence. We expect foraging may be temporarily disrupted if avoidance or displacement occurs, but we do not expect the loss of any breeding opportunities. Individuals engaged in travel or migration would continue with these activities, although potentially with a deflection of a few kilometers from the route they would otherwise pursue.

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

Stress is an adaptive response and does not normally place an animal at risk. Distress involves a stress response resulting in a biological consequence to the individual. The mammalian stress response involves the hypothalamic-pituitary-adrenal axis being stimulated by a stressor, causing a cascade of physiological responses, such as the release of the stress hormones cortisol, adrenaline (epinephrine), glucocorticosteroids, and others (Busch and Hayward 2009; Gregory and Schmid 2001; Gulland et al. 1999; St. Aubin and Geraci 1988; St. Aubin et al. 1996; Thomson and Geraci 1986). These hormones subsequently can cause short-term weight loss, the liberation of glucose into the blood stream, impairment of the immune and nervous systems, elevated heart rate, body temperature, blood pressure, and alertness, and other responses (Busch and Hayward 2009; Cattet et al. 2003; Dickens et al. 2010; Dierauf and Gulland 2001b; Elftman et al. 2007; Fonfara et al. 2007; Kaufman and Kaufman 1994; Mancia et al. 2008; Noda et al. 2007; Thomson and Geraci 1986). In some species, stress can also increase an individual's susceptibility to gastrointestinal parasitism (Greer et al. 2005). In highly-stressful circumstances, or in species prone to strong "fight-or-flight" responses, more extreme consequences can result, including muscle damage and death (Cowan and Curry 1998; Cowan and Curry 2002; Cowan and Curry 2008; Herraez et al. 2007). The most widely-recognized indicator of vertebrate stress, cortisol, normally takes hours to days to return to baseline levels following a significantly stressful event, but other hormones of the hypothalamic-pituitary-adrenal axis may persist for weeks (Dierauf and Gulland 2001a). Mammalian stress levels can vary by age, sex, season, and health status (Gardiner and Hall 1997; Hunt et al. 2006; Keay et al. 2006; Romero et al. 2008; St. Aubin et al. 1996). Stress is lower in immature right whales than adults are 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. 2012b). These levels returned to baseline after 24 hr of traffic resuming. As whales use hearing as a primary way to gather information about their environment and for communication, we assume that limiting these abilities would be stressful. Stress responses may also occur at levels lower than those required for TTS (NMFS 2006g). Therefore, exposure to levels sufficient to trigger onset of PTS or TTS are expected to be accompanied by physiological stress responses (NMFS 2006g; NRC 2003). As we do not expect individuals to experience TTS or PTS, (see *Marine mammals and threshold shifts*), we also do not expect any ESA-listed individual to experience a stress response at high levels. We assume that a stress response could be associated with displacement or, if individuals remain in a stressful environment, the stressor (sounds associated with the airgun, multibeam echosounder or sub-bottom profiler) will dissipate in a short period as the vessel (and stressors) transects away without significant or long-term harm to the individual via the stress response.

Exposure to loud noise can also adversely affect reproductive and metabolic physiology (Kight and Swaddle 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 hr 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).

Overall, sound can produce stress responses in mammals. The degree of this response (stress or distress) drives downstream physiological effects that can cause impacts ranging from normal physiological responses to lethal outcomes. We expect that exposure to loud sounds associated with the proposed airgun array will cause a stress response, but that this response will generally motivate individuals sufficiently to move away from the sound source and avoid more severe physiological responses.

Marine mammals and strandings. There is some concern regarding the coincidence of marine mammal strandings and proximal seismic surveys. No conclusive evidence exists to causally link stranding events to seismic surveys. For more discussion regarding marine mammal strandings related to anthropogenic acoustic sources, please see (NMFS 2013).

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

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

Some support has been found for fish or invertebrate mortality resulting from airgun exposure, and this is limited to close-range exposure to high-amplitudes (Bjarti 2002; Falk and Lawrence 1973; Hassel et al. 2003; Holliday et al. 1987; Kostyuchenko 1973; La Bella et al. 1996a; McCauley et al. 2000a; McCauley et al. 2000b; McCauley et al. 2003; Popper et al. 2005; Santulli et al. 1999). Nedelec et al. (2014) found boat noise playbacks to cause significantly higher levels of mortality in early life stage sea hares. Lethal effects, if any, are expected within a few meters of the airgun array (Buchanan et al. 2004; Dalen and Knutsen 1986). We expect fish to be capable of moving away from the airgun array if it causes them discomfort.

More evidence exists for sub-lethal effects on fishes and invertebrates. Several species at various life stages have been exposed to high-intensity sound sources (220-242 dB re: 1 μPa) at close distances, with some cases of injury (Booman et al. 1996; McCauley et al. 2003). TTS was not found in whitefish at received levels of ~175 dB re: 1 $\mu Pa^2 \cdot s$, but pike did show 10-15 dB of hearing loss with recovery within one day (Popper et al. 2005). Caged pink snapper have experienced PTS when exposed over 600 times to received seismic sound levels of 165-209 dB re: 1 $\mu Pa_{p\text{-}p}$. Exposure to airguns at close range were found to produce balance issues in exposed fry (Dalen and Knutsen 1986). Exposure of monkfish and capelin eggs at close range to airguns did not produce differences in mortality compared to control groups (Payne et al. 2009). Salmonid swim bladders (similar to the swim bladders of some marine mammal prey species) were reportedly damaged by received sound levels of ~230 dB re: 1 μPa (Falk and Lawrence 1973).

By far the most common response by fishes is a startle or distributional response, where fish react momentarily by changing orientation or swimming speed, or changing 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 after exposure to airguns (McCauley and Fewtrell 2013a). This effect generally persisted for several minutes, although subsequent exposures to the same individuals did not necessarily elicit a response (McCauley and Fewtrell 2013a). Startle responses were observed in rockfish at received airgun levels of 200 dB re: 1 $\mu Pa_{0\text{-}p}$ and alarm responses at >177 dB re: 1 $\mu Pa_{0\text{-}p}$ (Pearson et al. 1992). Fish also tightened schools and shifted their distribution downward. Normal position and behavior resumed 20-60 min after seismic firing ceased. A downward shift was also noted by Skalski et al. (1992) at received seismic sounds of 186–191 re: 1 $\mu Pa_{0\text{-}p}$. Caged European sea bass showed elevated stress levels when exposed to airguns, but levels returned to normal after three 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 2 hr following cessation of airgun activity. Whiting exhibited a downward distributional shift on exposure to 178 dB re: 1 $\mu Pa_{0\text{-}p}$ airgun sound, but habituated to the sound after one hour and returned to normal depth (sound environments of 185-192 dB re: 1 μPa) despite airgun activity (Chapman and Hawkins 1969). Whiting may also flee from airgun sound (Dalen and Knutsen 1986). Hake may redistribute downward (La Bella et al. 1996a). Lesser sand eels exhibited initial startle responses and upward vertical movements before fleeing from the survey area when approaching 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 μPa_{p-p} sound levels from an airgun (Thomsen 2002). In response to airgun activity cod and haddock likely leave seismic survey areas and estimated catchability decreases starting at received sound levels of 160–180 dB re: 1 μPa_{0-p} (Dalen and Knutsen 1986; Engås et al. 1996; Engås et al. 1993; Løkkeborg 1991; Løkkeborg and Soldal 1993; Turnpenny et al. 1994). Data collected by Lokkeborg et al. (2012) supports increased swimming activity in response to airgun exposure, as well as reduced foraging activity. Bass did not appear to leave during a shallow-water seismic survey with received sound levels of 163–191 dB re: 1 μPa_{0-p} (Turnpenny and Nedwell 1994). Similarly, European sea bass apparently did not leave their inshore habitat during a 4-5 month seismic survey (Pickett et al. 1994). La Bella et al. (1996b) found no differences in trawl catch data before and after seismic operations and echosurveys of fish occurrence did not reveal differences in pelagic biomass. However, fish kept in cages did show behavioral responses to approaching airguns. Schooling herring in a sea pen did not respond behaviorally to sounds in the 10 Hz-2 kHz range, although increases in cortisol and glucose indicated a stress response when killer whales sounds were played back (Handegard et al. 2013). No response was seen in a free-swimming school on the approach of a seismic airgun array (Pena et al. 2013). Passage of a seismic survey did not appear to alter the alter species richness of a demersal coral fish family compared to baseline conditions (Miller and Cripps 2013).

Squid responses to airguns have also been studied, although to a lesser extent than fishes. In response to airgun exposure, squid exhibited both startle and avoidance responses at received sound levels of 174 dB re: 1 $\mu Pa_{rms,}$ by first ejecting ink, and then moving rapidly away from the area (McCauley and Fewtrell 2013b; McCauley et al. 2000a; McCauley et al. 2000b). The authors also noted some movement upward. During ramp-up, squid did not discharge ink but alarm responses occurred when received sound levels reached 156–161 dB re: 1 μPa_{rms}. Guerra et al. (2004) suggested that giant squid mortalities were associated with seismic surveys based on coincidence of carcasses with the surveys in time and space, as well as pathological information from the carcasses. Sole et al. (2013) found damage to the statocysts of several squid species exposed to 50-400 Hz sounds with received sound levels of 157 ±5 dB re: 1 mPa with peak levels up to 175 dB re: 1 mPa). 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). Crayfish exposed to 100 Hz-25kHz signals in a tank showed blood and immune system changes as well as reduced aggression (Celi et al. 2013).

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

Marine mammal response to multibeam echosounder and sub-bottom profiler. We expect ESA-listed whales to experience ensonification from not only airguns, but also seafloor and ocean current mapping systems. Multibeam echosounder and sub-bottom profiler frequencies are much higher than frequencies used by all ESA-listed whales except blue, humpback, southern right 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 ESA-listed baleen whales, but it is assumed that they are most sensitive to frequencies over which they vocalize, which are much lower than frequencies emitted by the multibeam echosounder and sub-bottom profiler systems (Ketten 1997; Richardson et al. 1995c). Thus, if fin or sei whales are exposed, they are unlikely to hear these frequencies and a response is not expected.

Assumptions for blue, humpback, and sperm whale hearing differ from other ESA-listed whales. Humpback and sperm whales vocalize between 3.5-12.6 kHz and an audiogram of a juvenile sperm whale provides direct support for hearing over this entire range (Au 2000a; Au et al. 2006; Carder and Ridgway 1990; Erbe 2002a; Frazer and Mercado 2000; Goold and Jones 1995; Levenson 1974; Payne and Payne 1985; Payne 1970; Richardson et al. 1995c; Silber 1986; Thompson et al. 1986; Tyack 1983; Tyack and Whitehead 1983; Weilgart and Whitehead 1993; Weilgart and Whitehead 1997; Weir et al. 2007; Winn et al. 1970). MacGillivray et al. (2014) modeled sounds from a sub-bottom profiler (of lower frequency than that proposed for use in the proposed seismic survey) to be audible to humpback whales. 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 on exposure to 3.1-3.6 kHz sonar. Kremser et al. (2005) concluded the probability of a cetacean swimming through the area of exposure when such sources emit a pulse is small, as the animal would have to pass at close range and be swimming at speeds similar to the vessel. Sperm whales have stopped vocalizing in response to 6-13 kHz pingers, but did not respond to 12 kHz echo-sounders (Backus and Schevill 1966; Watkins 1977; Watkins and Schevill 1975). Sperm whales exhibited a startle response to 10 kHz pulses after 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 suggest a 12 kHz multibeam echosounder, similar in operating characteristics as that proposed for use aboard the *Revelle*, suggest that this sonar played a significant role in the mass stranding of a large group of melon-headed whales (Southall et al. 2013). The authors acknowledge that pathological data suggesting a direct physical effect is lacking, and the use of this type of sonar is widespread and common place globally although noted incidents like the Madagascar stranding can occur. 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 *Revelle*'s multibeam echosounder and that of the chase vessel. Although effects such as this have not been documented for ESA-listed species, the combination of exposure to this stressor along 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 NSF and ASC differ from sonars used during naval operations, which generally have a longer pulse duration and more horizontal orientation than the more downward-directed multibeam echosounder and sub-bottom profiler. The sound energy received by any individuals exposed to the multibeam echosounder and sub-bottom profiler sources during the proposed activities is lower relative to naval sonars, as is the duration of exposure. The area of possible influence for the multibeam echosounder and sub-bottom profiler is also much smaller, consisting of a narrow zone close to and below the source vessel. Because of these differences, we do not expect these systems to contribute to a stranding event.

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

7.3.2 Potential Responses of Sea Turtles to Acoustic Sources

Sea turtle response to airguns. 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, we do not expect either of these to occur to any sea turtle as a result of the proposed action.

Sea turtles and behavioral responses. As with listed whales, it is likely that sea turtles will experience behavioral responses in the form of avoidance. O'Hara and Wilcox (1990) found loggerhead sea turtles exhibited an avoidance reaction at an estimated sound level of 175–176 dB re 1 µPa $_{\text{rms}}$ (or slightly less) in a shallow canal. Green and loggerhead sea turtles avoided airgun sounds at received sound levels of 166 dB re 1 µPa and 175 dB re 1 µPa, respectively (McCauley et al. 2000a; McCauley et al. 2000b). Sea turtle swimming speed increased and becomes more erratic at 175 dB re 1 µPa, with individuals becoming agitated. Loggerheads also appeared to move towards the surface upon airgun exposure (Lenhardt 1994b; Lenhardt et al. 1983). However, loggerheads resting at the ocean surface were observed to startle and dive as active seismic source approached them (DeRuiter and Larbi Doukara 2012). Responses decreased with increasing distance of closest approach by the seismic array (DeRuiter and Larbi Doukara 2012). The authors developed a response curve based upon observed responses and predicted received exposure level. Recent monitoring studies show that some sea turtles move away from approaching airguns, although sea turtles may approach active seismic arrays within 10 m (Holst et al. 2006; LGL Ltd 2005a; LGL Ltd 2005b; LGL Ltd 2008; NMFS 2006e; NMFS 2006h).

Observational evidence suggests that sea turtles are not as sensitive to sound as are marine mammals and behavioral changes are only expected when sound levels rise above received sound levels of 166 dB re 1 µPa. This corresponds with previous reports of sea turtle hearing thresholds being generally higher than for marine mammals (DFO 2004). At 166 dB re 1 µPa, we anticipate some change in swimming patterns and a stress response of exposed individuals. Some turtles may approach the active seismic array to closer proximity, but we expect them to eventually turn away. We expect temporary displacement of exposed individuals from some portions of the action area while the *Revelle* 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.

7.4 Cumulative Effects

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

We expect that those aspects described in the *Environmental Baseline* will continue to impact ESA-listed resources into the near future. We expect climate change, habitat degradation, seismic surveys, vessel traffic, tourism, entrapment and entanglement, ship-strikes, and scientific research to continue into the future.

8 INTEGRATION AND SYNTHESIS OF EFFECTS

As explained in the *Approach to the assessment* section, risks to ESA-listed individuals are analyzed using changes to an individual's "fitness" – i.e., the individual's growth, survival, annual reproductive success, as well as lifetime reproductive success. When ESA-listed animals exposed to an action's effects are not expected to experience reductions in fitness, we do not expect the action to have adverse consequences on the viability of the population(s) those individuals represent or the species those populations comprise (Anderson 2000; Brandon 1978; Mills and Beatty 1979; Stearns 1992). As a result, if the assessment indicates that ESA-listed animals are not likely to experience reductions in their fitness, we conclude our assessment. If reductions in individuals' fitness are likely to occur, the assessment considers the risk posed to population(s) to which those individuals belong, and then to the species those population(s) represent. Figure 3 provides a conceptual organization as to how we considered fitness consequences.

ESA-Listed whales. The NSF proposes to fund the SIO to use the *Revelle* to conduct a seismic survey that could incidentally harass several ESA-listed marine mammal and sea turtle species; and the Permits and Conservation Division proposes to issue an IHA for marine mammals incidentally harassed during the survey. These species include blue whales, fin whales, humpback whales, sei whales, southern right whales, and sperm whales, all of whom are endangered throughout their ranges. It also includes green, hawksbill, leatherback, loggerhead South Pacific DPS, and olive ridley sea turtles, all of whom are listed as endangered throughout their range, except for green and olive ridley, which are listed as threatened.

The *status of listed resources* section identified commercial whaling as the primary reason for reduced populations, many of whom are a small fraction of their former abundance (Tables 3-7). Although large-scale commercial harvests no longer occur for these species, some harvests from subsistence and scientific research in regional and worldwide populations still occur. Other worldwide threats to the survival and recovery of ESA-listed whale species include altered prey base and habitat quality because 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 Southern 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 near future.

The *effects analysis* supports the conclusion of harassment to listed whales by proposed seismic activities. We expect up to 2 blue, 2 fin, 2 sei, 2 humpback, 2 southern right, and 10 individual sperm whales could be exposed to airgun sounds, which will provoke 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 percentage of the populations, with some individual re-exposure and reactions. These 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, southern right, or sei whales and, consequently, are not expected to have any direct effects on these species. However, blue, humpback, and sperm whales could hear sounds produced by these systems. Responses could include cessation of vocalization by sperm whales and/or movement out of the survey area by these species. Behavioral harassment caused by exposure to sound sources associated with the proposed seismic survey are expected to cause some individuals to cease these activities temporarily and possibly move out of the immediate area. We expect that individuals will resume foraging in a secondary location or reoccupy the habitat they were displaced from within hours to days. We do not expect these effects to have fitness consequences for any individual. The *effects analysis* also found that, although sperm whales may experience temporarily reduced feeding opportunities, this indirect effect would be transient and not reduce individual fitness of any whale from the survey or the IHA. Overall, we do not expect a fitness reduction to any individual whale. As such, we do not expect fitness consequences to populations or listed whale species as a whole.

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

These responses are expected from all individuals exposed and we do not expect a fitness consequence for any individual. As such, we do not expect survival or recovery consequences to populations or ESA-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, hawksbill sea turtles, leatherback sea turtles, loggerhead-South Pacific DPS sea turtles, and olive ridley sea turtles, which are either threatened or endangered. The *status of listed resources* section found that most sea turtle populations have undergone significant to severe reduction by human harvesting of both eggs and turtles, as well as severe bycatch pressure in worldwide fishing industries. As previously mentioned, the *cumulative effects* section identified actions in the *environmental baseline* to generally continue for the foreseeable future.

From the *effects analysis*, we expect that 3 green, 4 hawksbill, 7 leatherback, 4 loggerhead-South Pacific DPS, and 4 olive ridley 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 each population affected, but are expected to be exposed throughout the 1,066 km^2 area that will experience sound levels of at least 166 dB re: 1 µPa, based upon the amount of seismic survey trackline, source level of the airguns, and propagation modeling of sound dissipation. Based upon sighting and stranding data in New Zealand (Gill 1997a; Harley and Kendrick 2006; NZDOC 2014a; NZDOC 2014b; Pritchard 1982a), leatherbacks are expected to be exposed in larger numbers than any other species, followed by green sea turtles. Hawksbill, loggerhead, and olive ridley turtles may be rarely exposed, if at all. Data were not available to calculate the number of exposures, but we do not expect the number of ensonifications to alter critical life functions. We 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 survival or recovery of the populations these individuals belong or the viability of the species those populations comprise.

9 CONCLUSION

After reviewing the current status of the ESA-listed species, the environmental baseline within the action area, the effects of the proposed action and cumulative effects, it is NMFS' biological opinion that the proposed seismic survey off the New Zealand coast and NMFS' Permits and Conservation Division's issuance of an IHA action is not likely to jeopardize the continued existence of blue, fin, humpback, sei, southern right, and sperm whales as well as green, hawksbill, leatherback, loggerhead-South Pacific DPS, and olive ridley sea turtles. The proposed actions would have no effect on critical habitat.

10 INCIDENTAL TAKE STATEMENT

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

10.1 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 fin, blue, sei, humpback, southern right, 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.

The NMFS anticipates the proposed seismic survey along New Zealand is likely to result in the incidental take of ESA-listed species by harassment. We expect up to 2 blue, 2 fin, 2 sei, 2 humpback, 2 southern right, and 10 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 (

Table 21). Harassment is expected to occur at received levels above 160 dB re: 1 μPa. Additional exposures to the same individuals sufficient to elicit responses may also occur. We also expect individual green, hawksbill, leatherback, loggerhead-South Pacific DPS, and olive ridley 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 1,066 km^2 area along the New Zealand coast 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 *Revelle's* airgun array.

For all species of whales, 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.

Harassment of blue, fin, humpback, sei, southern right, and sperm whales exposed to seismic studies at levels less than 160 dB re: 1 μPa, or green, hawksbill, leatherback, loggerhead-South Pacific DPS, and olive ridley sea turtles exposed to seismic activities at levels less than 166 dB re: 1 μPa is not expected. If overt adverse reactions (for example, startle responses, dive reactions, or rapid departures from the area) by ESA-listed whales or sea turtles are observed at less intense levels than 160 dB or 166 dB re: 1 μPa for seismic activities, incidental take may be exceeded. The NSF, SIO, and NMFS's Permits and Conservation Division must contact the ESA Interagency Cooperation Division to determine whether reinitiation of consultation is required because of such responses.

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

10.2 Effect of the Take

In the accompanying Opinion, NMFS has determined that the amount of incidental take, coupled with other effects of the proposed actions, is not likely to jeopardize the continued existence of any ESA-listed species**.**

10.2.1 Reasonable and Prudent Measures

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

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

- The NSF and NMFS Permits Division must ensure that SIO implements and monitors the effectiveness of mitigation measures incorporated as part of the proposed authorization of the incidental taking of blue, fin, humpback, sei, southern right, and sperm whales pursuant to section 101(a)(5)(D) of the MMPA and as specified below for green, leatherback, loggerhead, and olive ridley 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.

10.2.2 Terms and Conditions

To be exempt from the prohibitions of section 9 of the ESA, the NSF, SIO, and Permits and Conservation Division must comply with the following terms and conditions, which implement the reasonable and prudent measure 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, SIO, and/or the Permits and Conservation Division fail to ensure compliance with these terms and conditions and their implementing reasonable and prudent measures, the protective coverage of section 7(o)(2) may lapse.

To implement the Reasonable and Prudent Measure, the NSF and the NMFS's Permits and Conservation Division shall ensure that:

- Mitigation and Monitoring Requirements:

1. Establish an exclusion zone[9] corresponding to the anticipated 180 dB re: 1 µParms isopleth for airgun array operations as well as a 160 dB re: 1 µParms buffer zone.
2. Use one NMFS-approved, vessel-based observer to watch for, and monitor, marine mammal or sea turtle species near the seismic source vessel during daytime airgun operations (nautical twilight-dawn to nautical twilight-dusk), and while the seismic array and streamers are being deployed and retrieved. When practical, vessel crew will also assist in detecting marine mammals or sea turtles. Protected species visual observer (observers) will have access to reticle binoculars (7 X 50 Fujinon), big-eye binoculars (25 x 150), optical range finders, and night-vision devices. Observer's shifts will last no longer than four hours at a time. When feasible, observers will also watch during daytime periods when the seismic system is not operating, for comparisons of animal abundance and behavior.
3. Record the following information when a marine mammal or sea turtle is sighted:
    e. 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.

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

    g. The data listed under b. would also be recorded at the start and end of each observation watch and during a watch whenever there is a change in one or more of the variables.
4. Use two observers to watch the entire exclusion zone, for at least 30 minutes (min) before starting the airgun (day or night). If observers find a marine mammal or sea turtle within the exclusion zone, SIO must delay the seismic survey until the marine mammal or sea turtle has left the area. If the observer sees a marine mammal or sea turtle that surfaces, then dives below the surface, the observer shall wait 30 min. During that time, if the observer sees no marine mammals or sea turtles they should assume that the animal has moved beyond the exclusion zone. If, for any reason, the entire radius cannot be seen for the entire 30 min (e.g., rough seas, fog, darkness), or if marine mammals or sea turtles are near, approaching or in the exclusion zone, the airguns may not be started up. If one airgun is already running at a source level of at least 180 dB re: 1 μPa_{rms}, SIO may start the second gun without observing the entire exclusion zone for 30 min prior, provided no marine mammals or sea turtles are known to be near the safety radius.
5. Apply a "ramp-up" procedure when starting up at the beginning of seismic operations or any time after the entire array has been shut-down for more than 15 min. This procedure means starting the smallest gun first and adding the second airgun in a sequence such that the source level of the array will increase in steps not exceeding approximately 6 dB per 5-min period.

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

During ramp-up, the two observers will monitor the 180 dB re: 1 µParms exclusion zone, and if marine mammals or sea turtles are sighted, a course/speed alteration or shut-down will occur as though the full array were operational.

6. Alter speed or course during seismic operations if a marine mammal or sea turtle, based on its position and relative motion, appears likely to enter the exclusion zone. If speed or course alteration is not safe or practical, or, if after alteration, the marine mammal still appears likely to enter the exclusion zone, further mitigation measures, such as shut-down, will be taken.
7. Shutdown the airguns whether marine mammals or sea turtles are detected within, approaching, or entering the exclusion zone. Airgun activity will not resume until the marine mammal or sea turtle has cleared the exclusion zone, which means it was seen leaving the exclusion zone, or has not been seen within the exclusion zone for 15 min (small odontocetes) or 30 min (sea turtles, mysticetes and large odontocetes).
8. Marine seismic operations may continue into night and low-light hours if segment(s) of the survey is started when the entire exclusion zone is visible and can be effectively monitored. Do not start 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 entire exclusion zone cannot be effectively monitored by the observer(s) on duty. To the maximum extent practicable, seismic airgun operations should be scheduled during daylight hours.
9. If the specified activity clearly causes any unanticipated cases of ESA-listed marine mammal or sea turtle injury or mortality resulting from these activities (e.g., ship-strike, gear interaction, and/or entanglement), SIO will cease operating seismic airguns and report the incident to NMFS's Office of Protected Resources at 301-427-8495, and/or by email to Cathy.Tortorici@noaa.gov of the ESA Interagency Cooperation Division, immediately.

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

   Airgun operation will then be postponed until NMFS is able to review the circumstances and then work with SIO to determine whether modifications in the activities are appropriate and necessary.
10. If SIO discovers an injured or dead marine mammal or sea turtle, and the lead observer determines both the cause of the injury or death is unknown and the death is relatively recent (i.e., in less than a moderate state of decomposition as described in the next paragraph), NSF

will immediately report the incident to the Chief of the Permits and Conservation Division, and Chief of the Endangered Species Act Interagency Cooperation Division, Office of Protected Resources, NMFS, at 301-427-8401 and 301-427-8495, and/or by email to Jolie.Harrison@noaa.gov, Howard.Goldstein@noaa.gov, and/or Cathy.Tortorici@noaa.gov. The report must include the same information as described above. Activities may continue while NMFS reviews the circumstances of the incident. NMFS will work with SIO to determine whether modifications in the activities are appropriate.

11. If SIO discovers an injured or dead marine mammal or sea turtle, and the lead visual observer determines that the injury or death is not associated with or related to the activities (e.g., previously wounded animal, carcass with moderate to advanced decomposition, or scavenger damage), SIO shall report the incident to the Chief of the Permits and Conservation Division, and Chief of the Endangered Species Act Interagency Cooperation Division, Office of Protected Resources, NMFS, at 301-427-8401 and 301-427-8495, and/or by email to Jolie.Harrison@noaa.gov, Howard.Goldstein@noaa.gov, and/or Cathy.Tortorici@noaa.gov., within 24 hr of the discovery. SIO shall provide photographs or video footage (if available) or other documentation of the animal sighting to NMFS. Activities may continue while NMFS reviews the circumstances of the incident.

In addition, the proposed incidental harassment authorization requires SIO to obey the following reporting requirements:

12. NSF is required to submit a report on all activities and monitoring results to the Office of Protected Resources, NMFS, within 90 days after the completion of the *Revelle*'s cruise. NSF shall provide this report to the ESA Interagency Cooperation Division. This report must contain and summarize the following information:
    h. 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 and sea turtle sightings.
    i. Species, number, location, distance from the vessel, and behavior of any marine mammals and sea turtles, as well as associated seismic activity (number of shutdowns), observed throughout all monitoring activities.
    j. An estimate of the number (by species) of marine mammals and sea turtles that:
        iii. Are known to have been exposed to the seismic activity (visual observation) at received levels greater than or equal to 160 dB re 1 microPa (rms) (marine mammals), 166 dB re 1 microPa (rms) (sea turtles), and/or 180 dB re 1 microPa (rms) during seismic airgun operations, with a discussion of any specific behaviors those individuals exhibited.
        iv. May have been exposed (modeling results) to the seismic activity at received levels greater than or equal to 160 dB re 1 microPa (rms) (marine mammals), 166 dB re 1 microPa (rms) (sea turtles), and/or 180 dB re 1 microPa (rms) with a discussion of the nature of the probable consequences of that exposure on the individuals that have been exposed.
    k. A description of the implementation and effectiveness of the:

v. Terms and conditions of the Opinion’s Incidental Take Statement.

vi. Mitigation measures of the IHA: For the Opinion, the report will confirm the implementation of each term and condition and describe the effectiveness, as well as any conservation measures, for minimizing the adverse effects of the action on listed whales and sea turtles.

11 CONSERVATION RECOMMENDATIONS

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

We provide the following conservation recommendations, which would inform us better in future consultations involving seismic surveys and the issuance of IHAs that may affect endangered large whales:

- The NSF should develop a more robust propagation model that incorporates environmental variables into estimates of how far sound levels reach from airgun sources.
- The NSF should promote and fund research examining the potential effects of seismic surveys on listed sea turtle species.

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

12 REINITIATION NOTICE

This concludes formal consultation on the proposed seismic survey to be carried out by SIO with funding from the NSF on board the *R/V Revelle* along New Zealand, and NMFS Permits and Conservation Division’s issuance of an IHA by the NMFS’ Permits and Conservation Division for the proposed studies pursuant to section 101(a)(5)(D) of the MMPA. As provided in 50 CFR §402.16, reinitiation of consultation will be required where discretionary Federal involvement or control over the action has been retained or is authorized by law, and: (1) if the amount or extent of incidental take is exceeded; (2) if new information reveals effects of the agency action that may affect ESA-listed species or critical habitat in a manner or to an extent not considered in this opinion; (3) if the agency action is subsequently modified in a manner that causes an effect to the ESA-listed species or critical habitat not considered in this opinion; or (4) if a new species is listed or critical habitat designated that may be affected by the action.

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