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

Biological Opinion

Agencies: The Bureau of Ocean Energy Management

The Bureau of Safety and Environmental Enforcement

Activities Considered: Programmatic Geological and Geophysical Activities in the Mid- and South Atlantic Planning Areas from 2013 to 2020

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

Approved by: Helen Golde for D.S. Wieting

Date: JUL 19 2013

Section 7(a)(2) of the Endangered Species Act (ESA) (16 U.S.C. 1531 *et seq.*) requires that each federal agency shall ensure that any action authorized, funded, or carried out by such agency is not likely to jeopardize the continued existence of any endangered or threatened species or result in the destruction or adverse modification of critical habitat of such species. When the action of a federal agency "may affect" a listed species or critical habitat that has been designated for such species, that agency is required to consult with NOAA's National Marine Fisheries Service (NMFS) and/or the U.S. Fish and Wildlife Service (USFWS), depending upon the listed resources that may be affected.

This ESA section 7 consultation considers BOEM's proposed geological and geophysical (G&G) activities in support of its oil and gas, renewable energy, and marine minerals programs in the Mid-and South Atlantic Planning Areas from 2013 through 2020, as described in BOEM's Draft Programmatic

Environmental Impact Statement (DPEIS) for Atlantic OCS Proposed Geological and Geophysical Activities in the Mid-Atlantic and South Atlantic Planning Areas (BOEM 2012a). These geological and geophysical (G&G) activities include various types of surveys (seismic and high resolution geophysical) and subsurface sampling activities in support of oil and gas exploration; detection of geohazards, archaeological resources and benthic communities; and the assessment of the suitability of seafloor sediments or evaluation of sand quantity and quality for beach nourishmentas well as bathymetric surveys used for monitoring sand resources before and after dredging. This consultation also considers monitoring and mitigation measures proposed to avoid or minimize adverse effects to listed species. BOEM's sister agency, the Bureau of Safety and Environmental Enforcement (BSEE), insures regulatory oversight and enforces compliance with monitoring and mitigation measures for oil and gas operations. Consequently, the action agencies for this consultation are BOEM and BSEE. As BOEM manages the exploration and development of the nation's offshore resources and has jurisdiction to authorize the G&G activities as described within this biological opinion (Opinion), BOEM is the lead agency for this consultation. The consulting agency is NMFS' Office of Protected Resources, Endangered Species Act Interagency Cooperation Division (Interagency Cooperation Division).

This document represents NMFS' biological opinion (Opinion) on the effects of the proposed action on endangered and threatened species and designated critical habitat, in accordance with section 7 of the ESA. This Opinion is based on information provided in BOEM's March 2012 Biological Assessment (BA; BOEM 2012b), DPEIS, published and unpublished scientific information on endangered and threatened species, scientific and commercial information such as reports from government agencies and peer-reviewed literature, regional experts in marine mammal biology, biological opinions on similar activities, and other sources of information.

Consultation History

On May 29, 2012, BOEM on behalf of itself and BSEE submitted a request for initiation of consultation on the proposed authorization of G&G activities under its oil and gas, renewable energy and marine minerals programs. Attached to the request was a BA dated May 22, 2012.

On August 6, 2012, the Interagency Cooperation Division transmitted a letter informing BOEM that NMFS had initiated consultation on the proposed authorization of G&G activities under its oil and gas, renewable energy and marine minerals programs. Attached to the initiation letter was a draft analysis plan for BOEM's review describing how NMFS intended to conduct its exposure analysis for the consultation. Throughout August and September, BOEM and the Interagency Cooperation Division exchanged emails regarding the draft analysis plan and information needs for the consultation.

On October 5, 2012, the Interagency Cooperation Division forwarded a draft description of the proposed action to BOEM for review. Also on October 5, 2012, NMFS asked for an extension to the consultation period due to the nature and complexity of the proposed action. On November 1, 2012, BOEM submitted its comments on the draft description of the proposed action. In its comments BOEM included changes to the proposed action by increasing surveys and subsurface sampling within its renewable energy program. Subsequently, NMFS requested an extension to the consultation period to fully analyze the additional survey and sampling effort. BOEM and NMFS exchanged information throughout November and into December. 1

On February 12, 2013, NMFS transmitted its draft biological opinion for BOEM and BSEE review. On March 13, 2013, BOEM submitted comments on the draft opinion to NMFS. The comments included language to further clarify BOEM's proposed action which includes authorizing certain surveys under the renewable energy and marine minerals programs within right whale critical habitat during right whale calving and nursing season. Because of BOEM's submission of its clarification, BOEM and NMFS agreed on March 26, 2013, to keep the consultation open while the agencies discussed the clarification.

On March 29, 2013, NMFS and BOEM discussed BOEM's comments on the draft opinion and BOEM's clarification and NMFS requested additional information on the clarification. On April 12, 2013, NMFS received the additional information requested along with revisions to the comments BOEM submitted to NMFS on March 13, 2012, on the draft opinion.

On April 17, 2013, BOEM submitted additional comments on the draft Opinion for NMFS' consideration.

On May 3, 2013, NMFS transmitted its revised draft biological opinion for BOEM/BSEE review. On May 9, 2013, BOEM submitted additional comments on the revised draft opinion. Additional discussions occurred between BOEM and NMFS from to May 9 to May 20, 2013, on terms and conditions for the incidental take statement.

Informal Consultation on Mid Atlantic Wind Energy Areas

On September 20, 2011, NMFS' Northeast Region issued a concurrence letter concluding informal consultation for BOEM's proposed issuance of leases and associated site characterization and site assessment activities as required for identifying suitable locations for wind energy facilities under its Renewable Energy Program. Activities under the proposed action include geophysical and geotechnical activities, wind resources assessments, biological resource surveys, cultural resources surveys, onshore activities (fabrication and staging of wind energy structures) and decommissioning of wind energy structures. Geophysical surveys employ single and multibeam depth sounders, side-scan sonar, chirp subbottom profilers and boomers while geotechnical surveys include cone penetrometer tests, vibracores and the installation of meteorological towers and buoys. All activities would occur within the Wind Energy Areas off New Jersey, Delaware, Maryland and Virginia.

As of the writing of this Opinion, none of the activities contained in the September 20, 2011, concurrence letter have taken place. During development of the proposed action for this G&G consultation BOEM re-examined the G&G acoustic propogation results for Delaware, Maryland and Virginia that were initially described and included as part of the proposed action under the 2011 informal consultation and compared those results with the results of the Acoustic Impact Model (AIM) modeling included in this G&G consultation. Due to the differences in the estimated exposure of marine mammals to some of the sound sources proposed for use, BOEM chose to update the assessment which formed the basis of the the 2011 informal consultation with the new information used for BOEM's assessment for this consultation. Given the updated assessment, this Opinion will only address the geophysical surveys using single and multibeam depth sounders, side-scan sonar, CHIRP subbottom profilers and boomers, geotechnical surveys using cone penetrometer tests and vibracores and the installation of buoys were analyzed during this consultation. As such, this Opinion supersedes the concurrence letter for those activities within Delaware, Maryland and Virginia. The September 20, 2011, concurrence letter remains in effect for the remaining activities (lease issuance, SAP approval, installation of met towers and biological surveys within New Jersey, Delaware, Maryland and Virginia Wind Energy Areas) contained therein, unless and until BOEM withdraws the proposed action or reinitiates consultation.

Scope of this Consultation

The Outer Continental Shelf Lands Act (OCSLA), as amended, mandates the Secretary of the Interior (Secretary), through BOEM, to manage the exploration and development of OCS oil, gas, and marine minerals (e.g., sand and gravel) and the siting of renewable energy facilities. The Energy Policy Act (EPAct) of 2005, Public Law (P.L.). 109-58, added Section 8(p)(1)(C) to the OCSLA, which grants the Secretary the authority to issue leases, easements, or rights-of-way on the OCS for the purpose of renewable energy development (43 U.S.C. § 1337(p)(1)(C)). The Secretary delegated this authority to BOEM (30 CFR 585).

BOEM, under the authority of OCSLA, as amended, regulates or otherwise requires the G&G activities that are the subject of this consultation within Federal waters. For the Atlantic, this includes waters between 3 nm (the limit for state waters) and 200 nm (the limit of the Exclusive Economic Zone (EEZ)). Some G&G activities, however, may occur within state waters and seaward of the EEZ beyond BOEM's authority. Activities that occur outside of BOEM's jurisdiction are still subject to the Marine Mammal Protection Act (MMPA) and actions authorized, funded or carried out by any Federal agency, including the issuance of MMPA authorizations, are subject to section 7 of the ESA. G&G activities that are the subject of this consultation and that may "take" marine mammals will require a MMPA Incidental Take Authorization from NMFS. At the time those authorizations are proposed for issuance, this consultation may be reinitiated.

There are currently no active oil and gas leases or oil and gas exploration, development, or production activities on the Atlantic OCS. Based on BOEM's 2012-2017 Proposed OCS Oil and Gas Leasing Program, no lease sales are proposed through 2012 and the earliest a lease sale may occur is in 2018, if proposed by BOEM's 2018-2022 Leasing Program.

Oil and Gas Program

G&G activities for the oil and gas program can take place anywhere over the shelf, slope, and abyssal plain at water depths ranging from 50 m to approximately 6,000 m except for areas governed by time/area closures as discussed in the *Mitigation and Monitoring* section later in this Opinion.

G&G activities can occur either pre-lease (before a federal lease for oil and gas has been issued under the Outer Continental Shelf Lands Act) or post-lease (after a lease has been issued). Typical pre-lease activities include deep penetration seismic airgun surveys to explore and evaluate deep geologic formations. Two-dimensional (2D) airgun surveys are designed to cover thousands of square miles or entire geologic basins as a means to geologically screen large areas. Three-dimensional (3D) surveys can consist of several hundred OCS blocks and provide much better resolution to evaluate hydrocarbon potential in smaller areas or specific prospects. OCS blocks are administrative boundaries (3x3 mi blocks in the Atlantic) that define geographic areas and identify federal land ownership to support offshore resource management). The 3D technology is also used to identify previously overlooked hydrocarbon-bearing zones and new productive horizons. However, because 3D modeling requires much denser data coverage (i.e., closer line spacing) than 2D seismic surveys, areas already covered using 2D techniques may be resurveyed. Other pre-lease surveys include largely passive data gathering methods such as electromagnetic, gravity, and magnetic surveys, as well as remote sensing surveys from aircraft and satellites.

Post-lease activities conducted by operators can include additional seismic airgun surveys, HRG surveys, and bottom sampling (including stratigraphic wells, shallow test wells, and geotechnical sampling). Examples of post-lease seismic airgun surveys include vertical seismic profiling (VSP) with geophone receivers placed in a wellbore. Four-dimensional (4D or time-lapse) surveys are 3D surveys repeated over time to monitor reservoirs during production. The HRG surveys are conducted on leases and along pipeline routes to evaluate the potential for geohazards, archaeological resources, and certain types of benthic communities. Geotechnical sampling is conducted to assess seafloor conditions with respect to siting facilities such as platforms and pipelines.

Seismic and HRG surveys are conducted using acoustic sources that emit acoustic energy pulses into the seafloor and receivers that record the returning acoustic signals. Acoustic sources consist of airguns, boomers, sparkers, side scan sonars, shallow and medium penetration subbottom profilers and single and multibeam depth sounders. Receiver arrays consist of hydrophones or geophones encased in plastic tubing and either towed behind the survey vessel, positioned on the seafloor as autonomous nodes or cables, or in rare instances spaced at various depths in vertically positioned cables depending on the type of survey conducted. Based on the requirements of the survey, one or more sources, vessels and receivers may be employed. Individual airguns are available with a wide range of chamber volumes, from under 5 in^3 to over 2,000 in^3 depending on survey requirements. Airgun arrays proposed for use in the Mid- and South Atlantic Planning Areas vary from about 90 in^3 (considered airgun HRG surveys) to approximately 5,400 in^3 (2D and 3D airgun seismic surveys) in volume with source levels up to about 230.7 dB re 1 μPa at 1m. Boomers, sparkers, side-scan sonars, subbottom profilers and single, swath and multibeam depth sounders, all considered HRG sources, may be used concurrently with airguns during seismic surveying or without airguns such as during non-airgun HRG surveys under the renewable energy and marine minerals programs.

Deep Penetration Seismic Airgun Surveys

2D Seismic Exploration Surveys.

2D seismic exploration surveys are generally conducted over large multi-block areas, whereas proprietary surveys usually cover only a few blocks. These seismic surveys are typically conducted with an airgun array consisting of three subarrays of six or seven airguns each and measuring approximately 12.5-18 m (41-60 ft) long and 16-36 m (52-118 ft) wide. The array is towed approximately 100-200 m (328-656 ft) behind the vessel between 5 and 10 m (16-33 ft) below the sea surface. Vessels typically travel between 4.5 and 6 kts (8.3 to 11 km/hour [hr]), respectively) and the airgun array is fired (emits pulses) every 37.5 m (123 ft) or every 16 seconds (s) at 4.5 kts and every 12 s at 6 kts.

When the airgun array is fired, an acoustic energy pulse is emitted and reflected or refracted back from the seafloor. These reflected/refracted signals are recorded by a 5-10 km (2.7-5.4 nmi) long, single streamer towed 100-200 m (328-656 ft) behind the array. A single ship generally 60-90 m (200-300 ft) long tows both the array and the streamer along a predetermined trackline for 12-20 hours (i.e., a distance of 100-166 km [54-90 nmi] for a vessel traveling at 4.5 kts [8.3 km/hr]) or 9-15 hours for a vessel traveling at 6 kts, depending on the size of the survey area to gather data for one trackline. Upon reaching the end of the trackline, the ship takes 2-3 hours to turn around and starts down another trackline. Although the airgun array is not fired during turns, operators fire a mitigation gun to maintain source levels at 160 dB re 1 µPa. The spacing between tracks is usually on the order of 2 km (1.1 nmi). This procedure takes place day and night and may continue for days, weeks, or months, depending on the size of the survey area.

3D Seismic Exploration Surveys.

3-D or narrow azimuth 3D surveys are typically conducted with two source arrays aligned in parallel with one another and towed 100-200 m (328-656 ft) behind a vessel. The two arrays are identical both to each other and to those used in the 2D surveys described above. The arrays are towed between 5 and 10 m (16-33 ft) below the sea surface and are fired in an alternate fashion. As with 2D surveys described above vessel speeds vary between 4.5 and 6 kts which varies the shot intervals; therefore, the first array fires at 16 s and the second array fires 16 s later with a ship speed of 4.5 kts or firing occurs every 12 s for a vessel traveling at 6 kts. Returning signals from the airgun array are recorded by 6-12 streamer cables 3-8 km (1.6-4.3 nmi) long and 600-1,500 m (1,969-4,922 ft) wide.

A single ship generally 80-90 m (262-295 ft) long tows both the array and the streamers along a predetermined trackline for 12-20 hours (i.e., a distance of 100-166 km [54-90 nmi] for a vessel traveling at 4.5 kts [8.3 km/hr]), depending on the size of the survey area. Upon reaching the end of the trackline, the ship takes 2-3 hours to turn around and starts down another trackline. During turns the airgun array is not fired, although a mitigation gun is fired to maintain source levels at 160 dB re 1 µPa. This procedure takes place day and night and may continue for days, weeks, or months, depending on the size of the survey area.

Wide Azimuth and Related Multi-Vessel Surveys.

New methods such as wide azimuth (WAZ), rich azimuth, multi-azimuth, and wide azimuth towed-streamer acquisition have emerged in the last few years to improve the data gathered by single vessel surveys. Wide, rich, and multi-azimuth surveys involve two to four source vessels (four is typical) in coordination with single or dual receiver vessels approximately 1,200-m (3,937 ft) apart in either a parallel or a rectangular arrangement. Receiver vessels may also deploy acoustic sources to improve data acquisition. Full azimuth (FAZ) or coil surveys can consist of a single source/receiver arrangement or a multi-vessel operation with multiple sources, with seismic data being acquired while the vessels follow a circular to spiral path. Because of vessel configuration and survey design, WAZ and FAZ surveys are able to record a broader range of source/receiver azimuths (directions) and a larger subsurface layer is surveyed. WAZ and FAZ surveys are considered 3D surveys utilizing multiple vessels, therefore, the airgun arrays and tow depths are identical to those described above for 3D surveys. Vessel configuration and alternating shot patterns during WAZ and FAZ surveys enables data acquisition over twice the area per trackline per pass than is normally covered during conventional 3D surveys. For these seismic surveys, airguns are fired every 13 s in an alternate fashion between all source vessels. As a result one quarter the number of shots is fired during WAZ surveys than those fired during a conventional 3D survey and there are no simultaneous shots. Data acquired from these surveys is volumetric as oppose to linear as in 2D surveys and is usually represented in OCS blocks or square kilometers.

Nodes and Ocean Bottom Cable Surveys. Ocean bottom cable surveys involve the use of remotely operated vehicles (ROVs) to deploy autonomous receiving units (nodes). With standard hydrophones, these surveys can be conducted in waters up to about 183 m (600 ft) but with newer technology surveys can reach waters depths of up to 2,500 m (8,200 ft) or more.

Autonomous nodes and ocean bottom cable surveys require the use of multiple ships (usually two ships for cable layout/pickup, one ship for recording, one ship for shooting, and two smaller utility boats). These ships are generally smaller than those used in 2- and 3D surveys using streamers, and the utility boats can be very small. Survey speed is typically 4.5 kts but can be as high as 6 kts. Once at the survey site, operations begin with the deployment of remotely operated vehicles (ROVs) to place nodes in equally spaced grids or by spooling cables off the back of the layout boat. Cable length is typically 4.2 km (2.3 nmi) but can be up to 12 km (6.5 nmi). Groups of seismic detectors (usually hydrophones and vertical motion geophones) are attached to the nodes and cable in 25-50 m (82-164 ft) intervals. Multiple nodes and cables are laid parallel to each other with a 50-m (164-ft) interval between cables. When the node or cable is in place, a ship towing a dual airgun array identical to those described for 3D surveys passes between the cables, firing every 25 m (82 ft) or 11 s for a vessel traveling at 4.5 kts (8.3 km/hr) or 8 s for a vessel traveling at 6 kts. After a source line is acquired, the source ship takes about 10-15 minutes to turn around and pass down between the next two nodes or cables.

A particular node or cable can lay on the bottom anywhere from 2 hours to several days, or if nodes or cables are needed for 4-D surveys they may lay on the bottom for years. When a node or cable is no longer needed to record seismic data, it is retrieved by the ROV or cable pickup ship and moved to the next recording position.

Vertical Cable Surveys. Vertical cable surveys, although uncommon, require two identically configured boats traveling at a speed of 4-5 kts (7.4-9.3 km/hr). Both boats are used initially to deploy the cables then one boat is used as a source boat and the other to recover and redeploy the cables. Vertical cables are deployed on two overlapping grids with vertical cables deployed every 2 km (1.1 nmi) on each grid. One grid is staggered relative to the other such that any one vertical cable is no more than 1.4 km (0.75 nmi) from its closest neighbor. Normally, 28 or 32 vertical cables are deployed at a time.

Cables are deployed by spooling off the back of the boat. At the bottom of each vertical cable is an anchor composed of 680 kilograms (kg) (1,500 pounds [lb]) of steel. The active section of the cable is 375 m (1,230 ft) long and contains 16 specially constructed hydrophones spaced 25 m (82 ft) apart. At the top are buoyant floats to keep the cable as vertical as possible.

Once the cables are in place, the survey begins. The source vessel is equipped with a dual airgun array identical to that described above for 3D streamer surveys. Shots are fired such that each vertical cable receives shots at a distance of 5 km (2.7 nmi) in all directions. This is accomplished by traveling down lines parallel to the grid of vertical cables. Survey speeds range from 4-5 kn (7.4-9.3 km/hr) and airguns are fired approximately every 10- 13 s. Once the source vessel shoots a line 1 km (0.54 nmi) beyond the first row of vertical cables, those cables are recovered and redeployed. Cables may be left in place for hours or days, depending on the size of the survey area and operating conditions. These surveys can be conducted in water depths up to about 2,500 m (8,200 ft).

4D (Time-Lapse) Surveys. 3D surveys may be repeated over oil and/or gas producing fields to characterize production reservoirs. These 4D or time-lapse surveys are 3D surveys repeated every 6 months to a year, typically, but can occasionally occur every 4 months. Time-lapse surveys can use either seismic streamers or ocean-bottom cables to house the seismic detectors. Time-lapse surveys are conducted similar to those described for 3D surveys when using seismic streamer cables or similar to ocean bottom cable surveys when using ocean-bottom cables. These surveys are conducted over much smaller areas than 2D surveys since the oil or gas field has already been located. An average survey takes 2-4 weeks and can cover 20 km^2 (5.8 nmi^2). When using fixed bottom cables, pre-survey operations are minimized since bottom cables are already deployed and all that has to be done is connect the fixed bottom cable to the recording instruments and start shooting.

Vertical Seismic Profile Surveys. Vertical seismic profiling is a technique carried out by using geophone receivers (sensor string) located on a cable and placed in a borehole at different depths to record acoustic signals from an external acoustic source near the wellbore (zero-offset VSP) or from a vessel at different distances from the wellbore (walk-away VSP). Zero-offset and walk-away VSP surveys are common during the development and production phases of oil and gas activity.

In all VSP surveys, sensors are lowered down a borehole before production tubing is placed in the wellbore or the well is abandoned. The sensors lowered down the borehole can be connected together in strings of 16-36 receivers spaced from 15-150 m (49-492 ft) apart, depending on the survey objective and other variables. After lowering the sensor string to the lowest portion of the borehole to be surveyed, the sensors are temporarily attached via a mechanical caliper that temporarily clamps to the side of the wellbore and seismic signals are recorded. Subsequently, the sensor string is repositioned and the next sets of seismic signals are recorded. Seismic sources used in VSP surveys are the same as those used in conventional 2D and 3D seismic airgun surveys.

Zero offset surveys are typically conducted using a single 3,000 in^3 airgun suspended approximately 10 m below the sea surface by a crane located on the deck of the drilling rig.

Walk-away surveys utilize a workboat with four to eight airguns towed 7-10 m (23-33 ft) below the surface. These surveys involve a source vessel firing at varying distances from the receivers within the borehole. The airgun arrays used for these surveys can vary from 1,000-5,000 $in.^3$ in volume, depending upon the survey objective. One version of walk-away surveys requires the source vessel to travel in a spiral track. The source vessel begins the spiral track at a distance of 200 m (656 ft) from the borehole and keeps the distance between spirals equal to the number of arrays times the array separation. Airgun arrays are fired in an alternating fashion with the first array firing followed by the second array 11-14 s later. At a typical vessel speed of 8.3-9.3 km/hr (4.5-5 kts), the distance between firings is between 28 and 36 m (92 and 118 ft). The source vessel continues firing on the spiral path out to a distance of up to 9 km (4.9 nmi). If the borehole sensor string needs to be raised to another level, the whole procedure is repeated.

Survey duration depends on the type of survey, objectives, cost of the drilling rig, and equipment used. A zero-offset survey can take less than a day to complete. A walk-away survey can be completed in less than one day or may require up to 10 days to complete, however, 30 percent of that time may be with the airguns in standby mode.

High-Resolution Geophysical Surveys

Industry conducts HRG surveys to investigate the shallow subsurface for geohazards and soil conditions and to identify potential benthic biological communities (or habitats). The HRG survey design, instrumentation and reports for oil and gas exploration will be guided by Notice to Lessees and Operators (NTL) issued for the Atlantic and similar to those issued for the Gulf of Mexico (i.e., NTL 2008-G05 for *Shallow Hazards Program and* NTL 2005-G07 for *Archaeological Resource Surveys and Reports*). HRG surveys for oil and gas exploration use airguns and several electromechanical acoustic sources including boomers, sparkers, side-scan sonars, shallow and medium penetration subbottom profilers and single, swath or multibeam depth sounders. Magnetometers, which passively measure the earth's magnetic field, are also employed during HRG surveys.

Electromechanical sources are considered mid- or high-frequency. BOEM expects that the frequency ranges used for non-airgun HRG surveys for oil and gas exploration are 200 Hz–16 kHz for the boomer; 50-500 Hz for the sparker; 100 kHz to 900 kHz for the side-scan sonar; 500 Hz to 24 kHz and 200 kHz for the chirp subbottom profiler; 3.5 kHz to 540 kHz for the single beam depth sounder; 100 kHz to 600 kHz for the swath depth sounder; and 240 kHz for the multibeam depth sounder. High-frequency electromechanical sources can be omni-directional or highly directive, with beam widths as narrow as a few degrees or less. Broadband source levels for electromechanical sources can range from 210 to 226 dB re 1µPa at 1 m.

Typical airgun HRG surveys commence with a single ship towing an airgun about 25 m (82 ft) behind the ship and a 600-m (1,969-ft) streamer cable with a tail buoy recording the acoustic signals. The ship travels at 3-3.5 kts (5.6-6.5km/hr) and the airgun is fired every 7-8 s (or about every 12.5 m [41 ft]). Typically one lease block is covered (usually 4.8 km on a side) per survey using a 30-m line spacing. Side-scan sonar, shallow and medium penetration subbottom profilers, and single, swath, or multibeam depth sounders are typically operated concurrently with the airgun array. Including line turns, the time to survey one block is about 36 hr; however, streamer and airgun deployment and other operations add to the total survey time.

3D high-resolution surveys use slightly larger (47 m vs. 37 m [154 ft vs. 121 ft]) ships since multiple streamers are towed. Up to six streamers 100-200 m (328-656 ft) long are used with a tri-cluster of airguns. With this system, 66 lines are necessary per block, which takes about 5 days to collect.

For post-lease engineering studies involving the placement of production facilities and pipelines in deep water, HRG surveys are often conducted with autonomous underwater vehicles (AUVs) equipped with a multibeam depth sounder, side-scan sonar, and a chirp subbottom profiler.

Electromagnetic Surveys

Electromagnetic surveys are often used in conjunction with seismic airgun surveys to help delineate potential oil and gas reservoirs. There are two electromagnetic techniques applicable to oil and gas exploration: Magnetotelluric (MT) surveys and controlled source electromagnetic (CSEM) surveys. MT surveys are passive measurement of the earth's electromagnetic fields; in this technique, no electrical currents are induced into the earth, but the receiver device detects the natural electrical and magnetic fields present in the earth.

The CSEM technique induces very low frequency electromagnetic signals into the upper layers of the seafloor via a towed dipole. A vessel tows the dipole approximately 1.5 to 2 kts (2.8-3.7 km/hr) along a towline usually within 50 m of the seafloor. In this technique, two cables (one a few hundred feet longer than the other) are joined together and towed by a ship. Attached to the end of each cable is a metal cylinder about 3 m (10 ft) long and 0.3 m (1 ft) in diameter. The towed dipole produces an electrical signal (typically less than 2 hertz [Hz]) that is continuous, not pulsed. These electrical signals are detected by an array of previously deployed receiver boxes 2-10 km (1.1-5.4 nmi) away from the source and arranged in a line or profile. The receiver boxes are attached to a degradable concrete anchor about 60 centimeters (cm) (24 inches [in]) on a side, 15 cm (6 in) high, and weighing about 136 kg (300 lb). Inside the receiver boxes are recording devices that record acoustic signals for a few days. When recording is finished, the transponder inside the receiver box is interrogated at a frequency of 35 kHz to 50 kHz frequency which triggers the release of the recording box from the anchor. The recording box floats to the surface for retrieval. These surveys last as little as several weeks to as much as one year as individual permits are valid for up to one year.

In MT surveys, ships are used to deploy and retrieve magnetotelluric recording devices. These devices are about 1.5 m (5 ft) high by 1 m (3 ft) on a side and are attached to a degradable concrete anchor like those used for CSEM surveys described above. The recording device has four arms sticking out from each side of the box with an electrode on each end. These arms are about 20 m (66 ft) long and made of polyvinyl chloride pipe. The recording box houses a magnetometer and a long-term recording device that allows the box to remain on the water bottom for days at a time. The recording box is released from the anchor using a transponder as described above for CSEM surveys and retrieved at the surface.

Table 1: Seismic Airgun and Electromechanical Surveys Proposed under the Oil and Gas Program for the Mid- and South Atlantic Planning Areas from 2013-2020.

Year	Mid-Atlantic Planning Area						South Atlantic Planning Area					
	2D (km)	3D (blocks) [a]	WAZ (blocks) [b]	HRG (line km)	VSP (line km)	CSEM (line km)	2D (km)	3D (blocks) [a]	WAZ (blocks) [b]	HRG (line km)	VSP (line km)	CSEM (line km)
2013	83,400	0	0	0	0	0	28,450	0	0	0	0	0
2014	160,950	0	0	0	0	0	56,900	0	0	0	0	0
2015	12,875	0	0	0	0	0	8,050	0	0	0	0	0
2016	64,375	400	0	0	0	3,220	48,300	300	0	0	0	1,600
2017	41,800	200	0	0	0	16,100	38,624	200	0	3,220	0	8,050
2018	16,100	200	100	3,220	0	32,200	32,200	200	100	32,200	0	9,650
2019	16,100	200	100	16,100	160	16,100	8,050	200	200	16,100	320	320
2020	800	300	200	64,375	320	32,200	800	300	200	40,250	480	320
TOTAL	**396,400**	**1,300**	**400**	**83,695**	**480**	**99,820**	**221,374**	**1,200**	**500**	**91,770**	**800**	**19,940**

Abbreviations: 2D = two-dimensional; 3D = three-dimensional; CSEM = controlled source electromagnetic; HRG = high resolution geophysical; VSP = vertical seismic profile; WAZ = wide azimuth.

[a] 3D surveys include ocean bottom cable and nodal surveys, vertical cable surveys, and 4D (time-lapse) surveys.

[b] WAZ estimates include coil shooting (exclusive to WesternGeco).

Deep Stratigraphic and Shallow Test Drilling

Although Continental Offshore Stratigraphic Test (COST) well or shallow test well drilling activity is infrequent and could occur anywhere within the Mid- or South Atlantic Planning Area, they are expected in the Blake Plateau region off South Carolina, within the South Atlantic Planning Area. A deep stratigraphic test well is defined as a well of at least 152 m (500 ft) in depth; otherwise it is classified as a shallow test well.

COST wells are a type of deep stratigraphic test well that is usually drilled to obtain information about regional stratigraphy, reservoir beds, and hydrocarbon potential. These wells are drilled away from potential petroleum-bearing features to minimize the chance of encountering oil or gas and are drilled using conventional, rotary drilling equipment from either a moored or dynamically positioned drilling rig. The choice of a moored or dynamically positioned drilling rig depends upon water depth, site-specific seafloor conditions and rig availability. Jack-up rigs typically are used in water depths less than 100 m (328 ft). Semisubmersibles are floating rigs that are used in depths ranging from 100 to 3,000 ft (328 to

9,843 ft) and can be either moored or dynamically positioned. Drillships are used in water depths greater than about 600 m (1,968 ft) and can also be moored or dynamically positioned (usually the latter).

COST wells drilled previously off the Atlantic seaboard ranged from 4,040 to 6,667 m below the seafloor in 41 to 819 m (136-2,686 ft) water depths. BOEM expects that a test program for gas hydrates in the Blake Plateau region of the South Atlantic Planning Area may occur with the request for either a COST well or a shallow test well (described below). Gas hydrate wells in the Gulf of Mexico ranged from 152 to 1,122 m (ft) below the seafloor.

Shallow test wells are drilled post-lease to allow operators to evaluate subsurface properties such as the presence of gas hydrates. Drilling would be done by conventional, rotary drilling equipment from a drilling barge or boat. Subsurface properties are evaluated by placing wireline testing equipment into the borehole. Wireline is a type of cabling used to lower equipment or measurement devices into the well to test well pressure and perform other well evaluations.

Drilling begins with a large-diameter surface hole jetted a few hundred meters into the seafloor. During drilling, drilling fluid and cuttings would be discharged, disperse in the water column, and accumulate on the seafloor around the well.

Bottom Sampling

Coring or grab sampling methods typically are used to obtain sediment samples for geological and/or geotechnical analyses. Geotechnical sampling and testing are used in engineering studies for placement of structures such as platforms and pipelines. Usually, a program of bottom sampling and shallow coring is conducted simultaneously using a small marine drilling vessel.

“Deep” geologic cores are obtained by standard rotary coring. The cores obtained by this method vary in diameter from 3-20 centimeters (cm) (1-8 inches [in.]) and can penetrate several hundred meters beneath the seafloor. Other methods used during geotechnical surveys include vibracorers, gravity corers, piston corers, box corers, and jet probes. Bottom sampling involves devices that penetrate only a few centimeters to several meters below the seafloor. Samples of surficial sediments are typically obtained by dropping a piston core or gravity core (“dart”), essentially a weighted tube, to the ocean floor and recovering it with an attached wire line. Grab sampling collects a sample of the topmost layers of the seabed by closing two steel clamshells together and cutting a sample from the soil.

Passive Remote Sensing Surveys

Remote sensing surveys use passive detection methods that do not involve a high-energy sound source. Gravity, gravity gradiometry, and marine magnetic surveys are remote sensing surveys typically conducted from ships. Aeromagnetic surveys are conducted by fixed wing aircraft and look for deep crustal structure, salt-related structure, and intra-sedimentary anomalies.

Gravity Surveys. Marine gravity data can be collected with instruments on the seafloor, in boreholes, or in helicopters, but is only expected to occur from ships and in conjunction with seismic surveys. Data grids for gravity surveys range from 1.6 by 8 km to 9.7 by 32 km (0.9 by 4.3 nmi to 5.2 by 17 nmi).

Gravity Gradiometry. Measuring the earth's gravity gradient is now possible with the release of Department of Defense (DOD) technology. The instrument is housed in a box located on a 60-m (197-ft) survey ship or fixed-wing aircraft. In shallow water, ships survey a 0.25- by 1-km (0.13- by 0.54-nmi) grid, and in deep water, a 1- by 2-km (0.54- nmi by 1.08-nmi) grid is used. Typically, a 20-block area is selected for surveying and a ship traveling at 11 kts (20 km/hr) can complete a survey in about 2 days. Gravity gradiometry surveys are also conducted with fixed-wing aircraft that fly at a speed of about 100 kts (185 km/hr) and altitudes of 80-100 m (262-328 ft).

Marine Magnetic Surveys. Marine magnetic surveys measure the earth's magnetic field for the purpose of determining structure and sedimentary properties of subsurface horizons. Magnetic surveys are also conducted to detect shipwrecks. These surveys are usually conducted in conjunction with a seismic survey, allowing the navigation information to be used for both surveys.

The magnetic sensor used for these surveys is housed in a cylindrical case measuring approximately 1 m (3 ft) long, 15-20 cm (6-8 in.) around and weighing about 14 kg (31 lb). The electronics package inside the case contains about 1 liter (L) (0.3 gallons [gal]) of chemically inert nontoxic fluid. The magnetic sensor package is towed approximately 100 m (328 ft) behind the seismic source array, although tow distances of 40 m (164 ft) or 150 m (492 ft) are also used. The sensor is typically towed at a depth of 3 m (10 ft) or in the case of surveys for archaeological resources, 6 m (20 ft) above the seafloor.

Aeromagnetic Surveys. Aeromagnetic surveys are conducted to look for deep crustal structure, salt-related structure, and intra-sedimentary anomalies by detecting the small magnetic fields of these structures that distort the earth's magnetic field. The magnetic field is measured by either a proton precision (using protons) or cesium vapor (using elemental cesium in vapor form) magnetometer mounted in a “stinger” projection, a rigid extension of the tail of the aircraft. Two magnetometers are also sometimes used to measure not only the total magnetic field but also the vertical gradient of the field. The surveys are flown by fixed-wing aircraft flying at speeds of about 250 km/hr (135 kts) to obtain a large number of readings over a relatively short time interval. Based on information for the northern Gulf of Mexico, most offshore aeromagnetic surveys are flown at altitudes between 61-152 m (200-500 ft) and collect 15,000-60,000 line km (9,320-37,282 line mi) of data. Line spacing varies depending on the objectives, but typical grids are 0.5 by 1.0 mi or 1.0 by 1.0 mi. A broad scale survey may be flown at higher altitudes (e.g., 305 m [1,000 ft]) and use wider line spacing (e.g., 4 by 12 mi or 8 by 24 mi).

Table 2: Proposed Level of Remote Sensing, Drilling and Bottom Sampling under the Oil and Gas Program in the Mid-and South Atlantic Planning Areas from 2013-2020.

Survey Type	Number of Sampling Events	Notes
Magnetotelluric Surveys	0-2 surveys	Hundreds to thousands of line km per survey, or ≤9 OCS blocks; 1-6 months per survey
Gravity and Magnetic Surveys (remote sensing)	0-5 surveys	Hundreds to thousands of line km per survey; 4-12 months per survey. Data typically acquired during seismic surveys (i.e., from ships)
Aeromagnetic Surveys (remote sensing)	0-2 surveys	Hundreds to thousands of line km per survey; 1-3 months per survey
Continental Offshore Stratigraphic Test (COST) Wells	0-3 wells	Penetration >150 m (500 ft). Requires an Environmental Assessment
Shallow Test Drilling	0-5 wells	Penetration <150 m (500 ft)
Bottom Sampling	50-300 samples	Mainly surficial and near-surface sediments; penetration <30 m (98 ft)

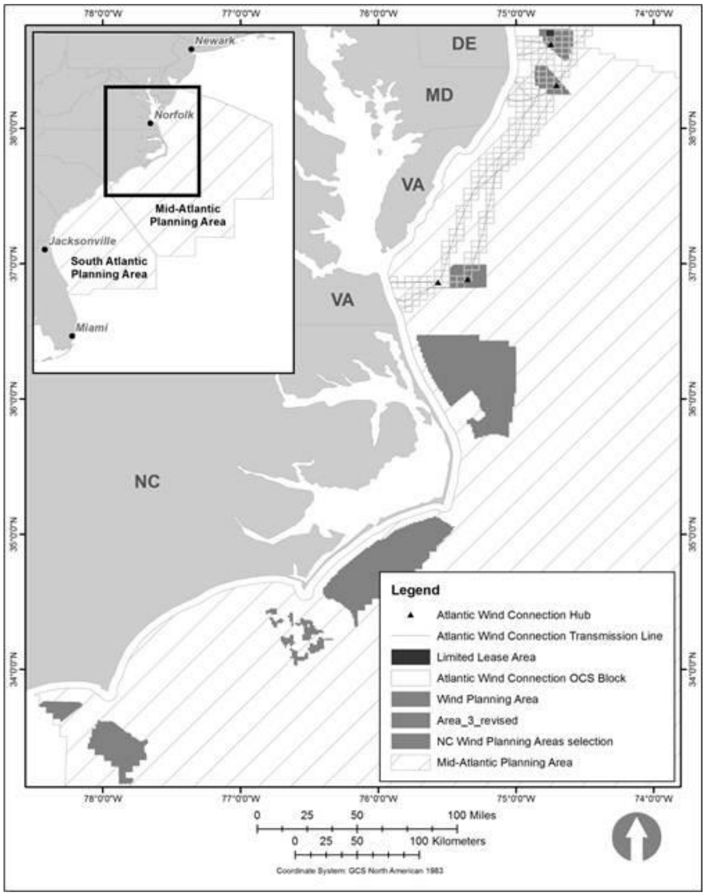
Renewable Energy Program

BOEM anticipates that developers will pursue construction and operation of wind energy facilities under its renewable energy program for the Mid- and South Atlantic Planning Areas. BOEM requires G&G surveys for site characterization and foundation studies for renewable energy projects. Several actions within the renewable energy program could trigger the requirement for G&G surveys: (1) BOEM's own planning and analysis; (2) unsolicited lease application to actualize plans for a renewable energy facility; (3) request for approval of a Site Assessment Plan (SAP) to construct a meteorological tower or install meteorological buoys on the leasehold; (4) request for approval of a Construction and Operations Plan (COP) which details the construction and operation of a wind energy facility on the lease; and (5) request for approval of a General Activities Plan (GAP) for rights-of-way (ROW) for installation of electrical cable in the seabed or for substations supporting an OCS wind energy facility on unleased OCS land or across land leased to a third party. Although BOEM does not directly permit or authorize G&G surveys for site characterization and foundation studies, BOEM does issue guidance specifying acceptable instrumentation and survey design parameters to complete the surveys as well as the specific reports required for approval of renewable energy activities (instrumentation and survey design parameters are discussed under the *High-Resolution Geophysical Survey* section below). BOEM will not approve SAPs, COPs, or GAPs without adequate G&G survey data. BOEM will conduct separate NEPA analyses and section 7 consultations for COP and GAP approval under its renewable energy program as required for the Mid- and South Atlantic Planning Areas. SAP approval in the mid-Atlantic WEAs was analyzed during informal consultation that resulted in a concurrence letter issued on September 20, 2011. BOEM will review each SAP and submit to NMFS it's determination as to whether the activity in the SAP is consistent with the activities and conditions outlined in the September 20, 2011, concurrence letter and this Biological Opinion, or whether additional section 7 consultation is necessary.

Generally, the outer limit for siting a wind energy facility is approximately 25 nmi (46.3 km) from shore or 100m (328 ft) water depth due to economic viability limitations. However, these facilities are typically sited around the 60-m isobath; therefore, any G&G activities associated with wind energy facility site characterization and foundation studies will occur within those limitations. Areas in which proposed site assessment activities for renewable energy development would occur are portions of the OCS along the U.S. eastern seaboard within identified Wind Energy Areas in the Mid-Atlantic Planning Area and offshore several states within the South Atlantic Planning Area (see also Figure 1 below):

- The Wind Energy Area (WEA) off Delaware is encompassed by 11 whole OCS blocks and 16 partial blocks (122 nmi^2, 91,093 ac) and lies between the incoming and outgoing shipping routes for Delaware Bay. The closest point to shore is approximately 10 nmi from Rehoboth Beach, DE.
- The WEA offshore Maryland is encompassed by 9 whole OCS blocks and 11partial blocks (94 nm^2, 79,706 ac) bounded on the western edge approximately 10 nmi from the Ocean City, MD, coastline and on the eastern edge approximately 27 nmi from the Ocean City, MD, coastline.
- The WEA offshore Virginia consists of 19 whole OCS blocks and 4 partial blocks (164 nmi^2, 138,788 ac) bounded on the western edge approximately 18 nmi from Virginia Beach and on the eastern edge approximately 37 nmi from Virginia Beach.
- For North Carolina BOEM has estimated that up to 354 OCS blocks or 2,015,428 ac will eventually undergo site assessments for potential wind energy development.
- For the states adjacent to the South Atlantic Planning Area (South Carolina, Georgia, and Florida), BOEM estimated that 30 lease blocks (204 nmi^2, 172,800 ac) offshore each of these states will eventually undergo site assessments for potential wind energy development. These blocks could lie anywhere off these states' coastlines.

Figure 1: Proposed Areas for Site Characterization and Assessment Activities within the Mid- and South Atlantic Planning Areas during 2013-2020.



For the renewable energy program HRG surveys, geotechnical surveys and deployment of bottom founded monitoring buoys are used for site characterizations for potential siting of renewable energy facilities. BOEM has also received an application for a right-of-way (ROW) grant for the Atlantic Wind Connection high voltage direct current cable and is assessing the site characterization survey work that would be needed to support the submittal of a GAP. BOEM will conduct separate NEPA and section 7 consultations upon submittal and consideration of a GAP for the transmission cable. The types of equipment employed and techniques used for site characterization activities associated with siting wind energy facilities and the Atlantic Wind Connection transmission cable are described below.

High-Resolution Geophysical Surveys

Lessees are required to collect HRG data to provide information on sub-seafloor conditions, shallow hazards, archaeological resources, and sensitive benthic habitats in a lease area and along transmission cable corridors. The area covered by an HRG survey encompasses all seafloor/bottom-disturbing activities (including the areas in which installation vessels, barge anchorages, and/or appurtenances may be placed) associated with construction, installation, inspection, maintenance, or removal of structures and/or transmission cables.

Equipment typically used in HRG surveys for renewable energy includes single beam, swath or multibeam depth sounders, magnetometers, side-scan sonars, shallow or medium penetration subbottom profilers, and boomers or sparkers. Air guns are not expected to be used. Specific requirements for HRG survey equipment follows:

- *Depth Sounders:* The depth sounder system should record with a sweep appropriate to the range of water depths expected in the survey area. Use of a multibeam bathymetry system is encouraged, particularly in areas characterized by complex topography or fragile habitats. Single beam and/or interferometric swath may also be used.
- *Magnetometers:* Magnetometer survey techniques should be capable of detecting and aiding the identification of ferrous, ferric, or other objects having a distinct magnetic signature. The magnetometer sensor should be towed as near as possible to the seafloor but should not exceed an altitude of greater than 6 m (20 ft) above the seafloor. The sensor should be towed in a manner that minimizes interference from the vessel hull and other survey instruments. The magnetometer sensitivity should be 1 gamma or less, and the background noise level should not exceed a total of 3 gammas peak-to-peak.
- *Side-Scan Sonars:* Recording should be of optimal quality (good resolution, minimal distortion) resulting in displays automatically corrected for slant range, lay-back, and vessel speed. The operator should use a digital dual-frequency side-scan sonar system with preferred frequencies of 445 and 900 kHz and no less than 100 and 500 kHz to record continuous images of the seafloor. The recorded data should be used to construct a mosaic to provide a true plan view that provides 100 percent coverage of the area of potential effect. The side-scan sonar sensor should be towed at a distance above the seafloor that is 10-20 percent of the range of the instrument.

- *Shallow and Medium Penetration Subbottom Profilers and Boomer:* A high-resolution chirp subbottom profiler is typically used to delineate near-surface geologic strata and features. A subbottom profiler system capable of achieving a vertical bed separation resolution of at least 0.3 m (1 ft) in the uppermost 15 m (49 ft) below the seafloor is recommended. The medium penetration boomer profiler system must be capable of penetrating greater than 10 m (33 ft) beyond any potential foundation depth, and the vertical resolution must be less than 6 m (20 ft).

The non-airgun HRG surveys are conducted from survey vessels fitted with equipment for deploying and handling geophysical systems. In nearshore waters, the surveys would be conducted by a single, small (~10-30 m [75-98 ft]) vessel moving at <5 kn (9.3 km/hr). Typically, a survey would be completed in 3-5 days, and depending on the location, the vessel may return to its shore base daily. Sites in deeper water may require larger vessels that operate 24 hr per day and can remain at sea for weeks. Survey vessels follow precise, pre-plotted lines so that the desired coverage of the seafloor is achieved. An integrated navigational system keeps track of the position and depth of the towed survey equipment.

BOEM recommends that the grid pattern for each survey should cover the area encompassing all anticipated physical disturbances including, but not limited to, the footprint of the renewable energy installation site and all areas associated with construction, installation, inspection, maintenance or removal of structures and/or transmission cables. Specific grid requirements are as follows:

- line spacing for all geophysical data for shallow hazards assessments (on side-scan sonar/all subbottom profilers) should not exceed 150 m (492 ft) throughout the area.
- line spacing for all geophysical data for archaeological resources assessments (on magnetometer, side-scan sonar, chirp subbottom profiler) should not exceed 30 m (98 ft) throughout the area. BOEM may require higher resolution surveys where necessary to ensure that site-specific actions comply with the National Historic Preservation Act.
- line spacing for bathymetric charting using multibeam technique or side-scan sonar mosaic construction should be suitable for the water depths encountered and provide both full coverage of the seabed plus suitable overlap and resolution of small discrete targets of 0.5-1.0 m (1.5-3 ft) in diameter.
- all track lines should run generally parallel to each other. Tie-lines running perpendicular to the track lines should not exceed a line spacing of 150 m (492 ft) throughout the survey area.

In addition, the geophysical survey grid for proposed transmission cable route(s) should include a minimum 300-m (984-ft) wide corridor centered on the transmission cable location(s). Line spacing should be identical to that noted above.

Geotechnical Surveys

Geotechnical surveys are used to aid in siting, design, construction, and operation of renewable energy facilities. Geotechnical surveys involve seafloor-disturbing activities such as cone penetrometer tests (CPTs), geologic coring, and grab sampling to determine the best foundations or anchoring systems for renewable energy facilities. These surveys occur after non-airgun HRG surveys have been conducted and BOEM clears the site for geotechnical work. Geotechnical surveys are typically completed within 3 days.

Sampling, including boring, should occur at every foundation location and at every kilometer of transmission cable route to shore, at a minimum. BOEM may request more than one deep geologic coring (cores taken from at least 10 m (33 ft) deeper than the proposed penetration depth of the foundation piles), with soil sampling and testing at each edge of the project area and within the project area to fully describe seafloor conditions relevant to facility design. BOEM may also request deep coring at each turbine foundation location.

Renewable energy geotechnical surveys are conducted from platforms such as a 20 m (65 ft) barge or ship. Platforms may require anchoring for sampling work or the use of vessels equipped with dynamic positioning capabilities. The area of seabed disturbed by individual sampling events (e.g., collection of a core or grab sample) is estimated to range from 1-10 m^2 (11-108 ft^2) and can occur in areas as small as 1/16 of lease block or over multiple lease blocks including cable routes to shore. Jack-up barges or spudded work barges may be used to deploy geotechnical survey equipment.

<u>Cone Penetrometer Tests (CPT)</u>. The CPT is widely used to gain information regarding soil composition and load bearing parameters without taking a soil sample. The CPT provides an *in situ* assessment of seabed soils based on the resistance of the soil to a cone-tipped probe, or penetrometer, as it is pushed into the seabed at a constant rate of penetration (about 2 cm/s [0.8 in./s]). Standard cones have a tip angle of 60 and a cross-sectional area between 5 and 20 cm^2 (0.8 and 3 in.^{2}). Electrical strain gauges within the cone assembly measure the soil's resistance through cone tip penetration and the resulting friction using a sleeve behind the tip. In a piezocone penetration test (PCPT), an additional parameter, soil pore water pressure, is measured via a porous element in the cone face or at the shoulder between cone tip and friction sleeve. The CPT device is connected to the vessel via a cable which is used to deploy and retrieve the device as well as transmit data for recording and analysis.

Geologic Coring. Geological cores are collected to describe surface and subsurface sediments for placement of renewable energy facilities. Standard rotary corers are employed to obtain 3-20 cm cores up to several hundred meters beneath the seafloor. Rotary corers are double or triple tube devices where the innermost tube acts as a core liner, the middle tube, if present, acts as a holder, and the rotating outer tube carries the hollow drill bit. As the bit cuts down through the soils and rock, the core created passes into the liner in a relatively undisturbed state.

At least one "deep" geologic coring (cores taken from at least 10 m (33 ft) deeper than the proposed penetration depth of the foundation piles) with soil sampling and testing at each edge of the project area and within the project area to fully describe seafloor conditions relevant to facility design is required. Other methods may be used during geotechnical surveys, including vibracorers, gravity corers, piston corers, box corers, and jet probes.

Grab Sampling. Grab samplers collect samples of the topmost layers of the seabed and benthic biota by bringing two steel clamshells together and cutting a bite from the soil. The grab sampler consists of two steel clamshells on a single or double pivot brought together either by a powerful spring or powered hydraulic rams operated from the support vessel. The grab is lowered to the seabed and activated either automatically or by remote control and the sample is raised to the vessel for examination. The sample is recovered to the ship for examination. Grab samplers used in renewable energy projects can retrieve samples of 0.35 m^3 (12.4 ft^3) or 700 kg (1,543 lb) at a rate of three to four grabs per hour. A typical hydraulic grab sampler can operate in water depths down to 200 m (656ft) and retrieve between three and four grabs per hour.

Bottom-Founded Monitoring Buoy Deployments

Instead of meteorological towers to characterize wind conditions for site suitability, BOEM anticipates that between 7 and 38 meteorological buoys anchored at fixed locations would gather wind and current data. Two types of buoys are likely to be employed: (1) discus buoys with circular hulls measuring between 10 and 12 m (33 and 39 ft) in diameter; and (2) boat-shaped hull buoys ("NOMAD") with an aluminum hull that provides long-term survivability in severe seas. Meteorological buoys can vary in height, hull type, and anchoring method; however, the largest meteorological buoys anticipated are similar to the spar buoys proposed offshore New Jersey by Garden State Offshore Energy. These buoys measure 30 m (100 ft) long and just over 2 m (6 ft) in diameter and are ballasted at one end to maintain tension and eliminate slack in the anchor chain. The buoy is equipped with an Acoustic Doppler Current Profiler which operates at frequencies between 190 kHz to 2MHz. Buoys are towed or carried aboard a vessel to the installation location. Once at the location site, the buoy would be either lowered to the water from the deck of the transport vessel or placed over the final location and dropped. Buoys placed in shallower waters may be moored with an all-chain mooring, while a larger discus-type buoy would use a combination of chain, nylon, and buoyant polypropylene materials. Buoy installation is estimated at 1 day including several hours to insure proper operation of all systems. Decommissioning of buoys is essentially the reverse of the installation process.

Atlantic Wind Connection Transmission Cable

BOEM has received an application for a right-of-way grant for a high voltage direct current cable that would inter-connect future offshore wind energy facilities. BOEM is assessing the site characterization survey work (non-airgun HRG surveys and geotechnical work) that would be needed to support the future submittal of a GAP. The proposed cable route, and thus survey track, would be 1,320 km (820 mi) running from northern New Jersey through Delaware to Virginia. The ROW is approximately 1,320 km (820 mi) in length. Under BOEM's regulations, a right-of-way is 61 m (200 ft) in width, though the developer may elect to perform non-airgun HRG and geotechnical surveys on a somewhat wider area to facilitate rerouting and placement of a dozen offshore platforms (substations).

Table 3: Projected Levels of G&G Activities for Renewable Energy Site Characterization and Assessment in the Mid- and South Atlantic Planning Areas, 2013-2020.

Renewable Energy Area	HRG Surveys[a] (max km/hours)	Geotechnical Surveys[b]			Bottom-founded Monitoring Buoys (min-max)	Timing
		CPT (min-max)	Geologic Coring (min-max)	Grab Samples (min-max)		
Delaware	14,880/2,410	224–720	224–720	224–720	1–2	2013-2016
Maryland	13,030/2,110	196–630	196–630	196–630	1–6	2013-2017
Virginia	18,400/2,980	266–855	266–855	266–855	1–6	2013-2017
North Carolina	327,850/53,150	4,956–15,930	4,956–15,930	4,956–15,930	1–20	2013-2020
Mid-Atlantic Subtotal	**374,160/60,650**	**5,642-18,135**	**5,642-18,135**	**5,642-18,135**	**5–40**	**2013-2020**
South Carolina	27,830/4,510	420–1,350	420–1,350	420–1,350	1–6	2013-2017
Georgia	27,830/4,510	420–1,350	420–1,350	420–1,350	1–6	2013-2018
Florida	27,830/4,510	420–1,350	420–1,350	420–1,350	1–6	2013-2018
South Atlantic Subtotal	**83,490/13,530**	**1,260–4,050**	**1,260–4,050**	**1,260–4,050**	**3–18**	**2013-2018**
Atlantic Connection Transmission Cable	6,600/820	12–24	12–24	12–24	0	2013-2020
TOTAL	**464,250/75,000**	**6,914–22,209**	**6,914–22,209**	**6,914–22,209**	**8–58**	**2013-2020**

Abbreviations: HRG = high-resolution geophysical; CPT = cone penetrometer test.

[a] HRG survey effort per block was assumed to be 925 km (500 nmi), requiring 150 hr to complete. Added 80 km (43 nmi) and 10 hr for surveying one transmission cable route for each state (except NC which has 5 separate planning areas and could have 5 independent transmissions cables). For the Atlantic Wind Connection transmission cable, the proposed route length of 1,320 km (820 mi) was multiplied by 5 km per kilometer of route.

[b] Geotechnical survey effort was estimated to be 14-45 sampling locations per block based on the potential range of wind turbine densities per block (assuming one sampling location per turbine location). For the Atlantic Wind Connection transmission cable, assumed up to 12 substations with one or two sampling locations per substation.

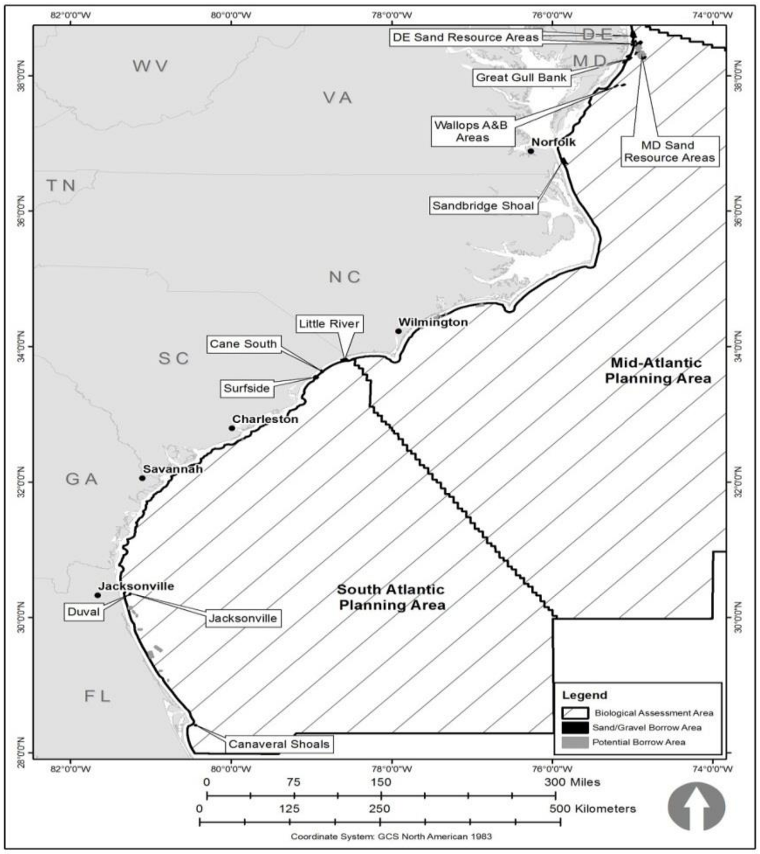
Marine Minerals Program

The marine minerals program manages the sand and gravel resources on the OCS through authorizing G&G prospecting for sand resources, as well as authorizing G&G surveying for borrow area monitoring before and after dredging and construction. In addition to identifying OCS sand resources, G&G surveys are conducted to identify any environmental resources such as sensitive benthic habitats, cultural resources, and shallow hazards that may exist in potential sand and gravel resource (borrow) areas as well as to further characterize the volume and quality of a prospective borrow area. Although the exact locations for G&G surveys for sand and gravel identification and assessment activities are not known at this time, BOEM has determined using existing borrow site information and future use projections that much of the G&G survey activity is expected to occur within existing borrow areas offshore the Mid-and

South Atlantic, with probably 40-50 percent repeated use of the Sandbridge Shoal, Virginia, and Canaveral Shoals, Florida and Jacksonville, Florida borrow areas (See Figure 2). New borrow areas along any coastal state from Delaware to Florida (except Georgia) may be discovered and used as well. Whether borrow areas accessed are existing or new, current dredging technology effectively limits all borrow areas to less than 30 m (98 ft) water depths and all existing areas are within 10-30 m depths (33 to 98 ft). When existing sites are reused in the future, additional G&G surveying, with the exception of operational or monitoring bathymetric surveys, is not usually required, but may be conducted if needed to detect archaeological resources and other bottom obstructions and to sample the area to see if it can provide the quantity and quality of sand required.

Two general types of G&G surveys are expected to be conducted in support of marine mineral uses, non-airgun HRG surveys and geotechnical surveys. Non-airgun HRG surveys involve single beam, swath interferometric, or multibeam depth sounders, side-scan sonar, a magnetometer and a chirp subbottom profiler or a boomer. Geophysical surveys involve seafloor disturbing activities such as coring and grab sampling. These surveys, although much smaller in scale (usually 1-3 OCS blocks per survey and usually 1-2 days in duration), are conducted similarly to those under the renewable energy program and are described in more detail below.

Figure 2: Sand and Gravel Borrow Areas Likely Used Along the Mid- and South Atlantic Coasts From 2013-2020.



Non-Airgun High-Resolution Geophysical Surveys

Prospecting and Pre-Lease Geophysical Surveys. Prospecting and pre-lease surveys usually involve a single vessel (<10-20 m [35-65 ft] traveling less than 5 kts (9.3 km/hr, typically 8.3 km/hr) while deploying single beam, swath, or multibeam depth sounders, side-scan sonar, a magnetometer and a chirp subbottom profiler or a boomer. Surveys can be conducted over reconnaissance areas (generally 1-3 OCS blocks, 23.3- 69.9 km^2) to locate prospective sand resources or over prospective borrow areas (generally 3-10 km^2 [300-1,000 ha or 741-2,471 ac]) to identify promising borrow targets. Prospecting surveys may be conducted at line spacing between 150 and 600 m (492 and 1,969 ft). During the reconnaissance phase, limited geotechnical sampling often occurs at a geographic point along already acquired seismic lines to validate geophysical data interpretations. Pre-lease (or design-level) non-airgun HRG surveys may be used to prepare a dredging plan or to refine the borrow area and/or determine horizontal and vertical continuity of sedimentary units (in which case, the survey may be subject to BOEM authorization). These surveys may be conducted at 15-50 m (49-164 ft) line spacing. Individual prospecting and pre-lease surveys are usually completed within 1-5 operational days, depending on the weather and sea state and survey footprint.

On-Lease Geophysical Surveys. BOEM uses on-lease non-airgun HRG surveys to monitor the location and volume of sand dredging, the structure of sand bodies and borrow areas and to insure observance of exclusion zones around cultural resources. About 1 mi^2 (259 ha or 640 ac) is surveyed at the borrow area or a sub-section of the borrow area; therefore these surveys are usually completed within 1-2 days. Bathymetric surveys using high frequency single beam, swath, or multibeam sonar are the most common type of geophysical surveys but side-scan sonar may also be used if sensitive cultural or benthic resources cannot be avoided during dredging.

Table 4: Proposed Levels of High-Resolution Geophysical Surveys for OCS Sand Borrow Projects In the Mid- and South Atlantic Planning Area under the Marine Mineral Program From 2013-2020.

Year	Project	State	Cycle Volume (cubic yd)	Depth (m)	Distance Offshore (km)	Prospecting HRG[a] (line km)		Pre-Lease HRG[a] (line km)		On-Lease HRG[b] (line km)	
						(lower bound)	(upper bound)	(lower bound)	(upper bound)	(lower bound)	(upper bound)
Mid-Atlantic Planning Area											
2013	Wallops Island	VA	3,200,000	9-24	18-20	0	0	0	0	100	501
	Fort Story/Dam Neck	VA	1,000,000	9-20	5	0	0	0	0	31	156
	Sandbridge	VA	2,000,000	9-20	5	0	0	0	0	63	313
2014-2016	Rehoboth/Dewey	DE	360,000	9-20	5	26	642	47	235	11	56
	Bethany/S. Bethany	DE	480,000	9-20	5	34	856	63	313	15	75
	Atlantic Coast of Maryland	MD	800,000	12-16	12-16	0	0	104	522	25	125
	Wallops Island	VA	806,000	9-24	18-20	0	0	0	0	25	126
	Sandbridge	VA	2,000,000	9-20	5	0	0	0	0	63	313
	West Onslow/North Topsail	NC	866,000	13-15	6-9	0	0	0	0	27	135
	Bogue Banks	NC	500,000	13-15	3-5	0	0	65	327	16	78
2017-2020	Rehoboth/Dewey	DE	360,000	9-20	4.8	0	0	0	0	11	56
	Bethany/S. Bethany	DE	480,000	9-20	4.8	0	0	0	0	15	75
	Atlantic Coast of Maryland	MD	800,000	12-16	12-16	0	0	0	0	25	125
	Surf City/North Topsail	NC	2,640,000	12-15	5-8	0	0	0	0	83	413
	Wrightsville Beach	NC	800,000	N/A	N/A	34	856	104	522	25	125
South Atlantic Planning Area											
2013	Patrick Air Force Base	FL	310,000	3-14	3-8	0	0	0	0	10	49
2014-2016	Grand Strand	SC	2,300,000	7-13	4-7	0	0	0	0	72	360
	Brevard County North Reach	FL	516,000	3-14	3-8	0	0	0	0	16	81
	Brevard County Mid-Reach	FL	900,000	3-15	3-8	0	0	0	0	28	141
	Brevard County South Reach	FL	850,000	3-16	3-8	0	0	0	0	27	133
2017-2020	Folly Beach	SC	2,000,000	12-14	5	0	0	261	1306	63	313
	Duval County	FL	1,500,000	14-19	10-11	0	0	0	0	47	235
	St. Johns	FL	N/A	N/A	3-6	N/A	N/A	N/A	N/A	N/A	N/A
	Flagler	FL	N/A	N/A	3-5	N/A	N/A	N/A	N/A	N/A	N/A
	TOTALS										
2013-2020	Mid-Atlantic Planning Area		17,092,000			94	2,354	383	1,919	535	2,672
	South Atlantic Planning Area		8,376,000			0	0	261	1,306	263	1,312
	Unknown Projects in Mid- and South Atlantic Planning Areas		8,000,000	N/A	N/A	34	856	209	1,045	125	626
	Mid- and South Atlantic Planning Areas		33,468,000			128	3,210	853	4,270	923	4,610

[a] Prospecting and pre-lease HRG involves the use of subbottom profiler, side-scan sonar, bathymetry (depth sounders), and magnetometer.
[b] On-lease typically involves only a bathymetry survey (depth sounders).

Geotechnical Surveys

Nearly all geotechnical sampling occurs from either stationary vessels less than approximately 20 m (65 ft) in length or from work barges towed into place. Some operational platforms require anchoring for brief periods with small anchors; however, approximately 50 percent of deployments for this sampling work could involve a boat having dynamic positioning capability. Jack-up barges and spudded work barges are seldom used. Surveys typically last 3 days or less and disturb between 1 and 10 m^2 (11 and 108 ft^2) of seabed during individual sampling events (e.g., collection of a core or grab sample).

Geotechnical sampling is most frequently done in connection with reconnaissance geophysical surveying and includes vibracoring, geologic (standard rotary) coring, and /or grab sampling. Although vibracoring is the most likely technique used, other sampling methods such as piston or box coring and jet probes are also used as part of geotechnical surveys.

Vibracoring. Vibracoring is usually performed with a 7-cm (2.8-in.) diameter core barrel mounted on a platform or tripod support assembly and can penetrate sediments in the upper 15 m (50 ft). Vibracores are often diver assisted. The vibracore barrel is vibrated, facilitating its penetration into the sediment. A typical vibracore survey will obtain 15-25 cores, approximately 6 m (20 ft) deep in a 1 mi^2 (640-ac or 259-ha) area. Vibracores are performed from a small gasoline powered vessel, usually less than 14 m (45 ft).

Geologic coring and grab sampling. These techniques are the same as those described earlier (see Oil and Gas section). Core samples vary in diameter from 3-20 cm (1.2-7.9 in.) and can penetrate several hundred meters beneath the seafloor. Because of the significantly greater expense, only one to two geologic cores would typically be drilled in a 1-mi^2 (640-ac or 259-ha) area. Grab sampling penetrates from a few inches to a few feet below the seafloor and typically involves 30-40 grabs in the area of interest.

Table 5: Proposed Levels of Geotechnical Surveys for OCS Sand Borrow Projects in the Mid- and South Atlantic Planning Areas under the Marine Minerals Program From 2013-2020.

Type of Geotechnical Sampling	Number of Deployments	Number of Samples Per Deployment	Number of Samples
Vibracoring	6-24	15-25	90-600
Geologic coring	1-4	1-2	1-8
Grab sampling	2-8	30-40	60-320

Table 6: Total G&G Activities within the Oil and Gas, Renewable Energy and Marine Minerals Programs from 2013 through 2020.

Activity Type	Purpose	Number of Events or Level of Effort	Primary Platform and Size	Scale of Activity	Penetration Depth	Approximate Duration/ Event	Shore Base[a]	Service Vessel	High-Energy Sound Source(s)	Bottom Area Disturbed
Oil and Gas Exploration										
2D Seismic Survey	Identify geologic structure	1-10	1 ship, ~100 m	617,775 line km	kms to 10s of kms	2-12 months	0 to 1	0 to 1	Airgun array	None
3D Seismic Survey	Identify geologic structure	5-10	1-2 ships, ~100 m	2,500 OCS blocks	kms to 10s of kms	4-12 months	0 to 1	0 to 1	Dual airgun array	None
3D WAZ and 3D FAZ Coil	Better define complex geologic structure	1-2	4 ships, ~100 m	900 line km	kms to 10s of kms	1 year	0 to 2	1 to 2	4 x arrays	None
Vertical Seismic Profiling	Calibrate seismic with known geology	3-8	1 ship, ~30 m	1,280 line km	100s to 1,000s of m	3-4 days	1	None	Single airgun	
High-Resolution Seismic Survey	Shallow hazards assessment and archaeological determinations	10-20	1 ship, ~30 m	175,465 line km	10s to 100s of m	3 days – 1 week	1	None	• 1-2 airguns • Boomer or chirp subbottom profiler • Side-scan sonar • Multi-beam depth sounder • Sparker	None
3D Controlled Source Electromagnetic	Optimize reservoir production	0-2	1 ship, ~20-100 m	119,760 km	3-5 km	1-6 months	0 to 1	0 to 1	None	Anchors with bottom receivers, <1 OCS block
Magnetotelluric Survey	Optimize reservoir production	0-2	1 ship, ~20-100 m	100s to 1,000s of line kms; or ≤9 OCS blocks	3-5 km	1-6 months	0 to 1	0 to 1	None	Anchors with bottom receivers, <1 OCS block
Gravity and Magnetic	Passive measurement, gravity and magnetic fields	0-5	Acquisition with seismic typical	100s to 1,000s of line kms	kms to 10s of kms	4-12 months	0 to 1	0 to 1	None	None
Aeromagnetic	Passive measurement, magnetic fields	1-2	1 aircraft	100s to 1,000s of line kms	kms to 10s of kms	1-3 months	0 to 1	0	None	None
COST Well	Test drilling outside of lease program	0-3 well	Platform or drillship, ~100 m	<1/16 OCS block	≥150 m	5-30 days	0 to 1	0 to 2	None	≤2 ha per well
Shallow Test Drilling	Test drilling outside of lease program	0-5 wells	Platform or drillship, ~100 m	<1/16 OCS block	<150 m	5-30 days	0 to 1	0 to 2	None	≤2 ha per well
Bottom Sampling	Extract sediment core	50-300	1 barge or ship, ~20 m	<1/16 OCS block	<300 m	<3 days	0 to 1	None	None	~10 m^2, per sample

Activity Type	Purpose	Number of Events or Level of Effort	Primary Platform and Size	Scale of Activity	Penetration Depth	Approximate Duration/ Event	Shore Base[a]	Service Vessel	High-Energy Sound Source(s)	Bottom Area Disturbed
Renewable Energy										
Non-airgun High-Resolution Geophysical Survey	Shallow hazards assessment and archaeological determinations	1 or more surveys per state	1 ship, ~20-30 m	Each survey ≥1/16 OCS block plus cable route to shore; Total 464,250 line km	Surficial to 10s to 100s of meters	3 days – 1 week	1	None	• Boomer or chirp subbottom profiler • Side-scan sonar • Single, swath,and multi-beam depth sounders	None
Cone Penetrometer Test	Measure sediment engineering properties	6,914-22,209	1 barge or ship, ~20 m	≥1/16 OCS block or along cable route to shore	<10 m	<3 days	1	None	None	~10 m^2 per sample
Geologic Coring	Extract sediment core	6,914-22,209	1 barge or ship, ~20 m	≥1/16 OCS block or along cable route to shore	<300 m	<3 days	1	None	None	~10 m^2 per sample
Grab Sampling	Collect sediment and benthic fauna	6,914-22,209	1 barge or ship, ~20 m	≥1/16 OCS lock or along cable route to shore	<1 m	<3 days	1	None	None	~10 m^2 per sample
Bottom-Founded Monitoring Buoy	Measure ocean and meteorological conditions	8-58	1 barge or ship, ~20 m	≥1/16 OCS block	Surficial	<3 days	1	None	None	~75 m^2 per buoy
2D or 3D Deep Penetration Seismic[c]	Evaluate formation for carbon sequestration	0 to 1 survey	1 ship, ~100 m	<1 OCS block	km to 10s of km	1 - 30 days	1	0-2	Airgun array or dual array	None
Marine Minerals										
Non-airgun High-Resolution Geophysical Survey	Shallow hazards assessment and archaeological determinations	10-40 surveys, 9-21 wks	1 ship, ~10-30 m	~1,904-12,090 line kms; or 1-4.5 OCS blocks	10s to 100s of m	1 day - 1 week	1	None	• Boomer or chirp subbottom profiler • Side-scan sonar • Single, swath, and multi-beam depth sounders	None
Vibracoring	Extract sediment core	6-24 events (90-600 cores)	1 barge or ship, ~20 m	≥1/16 OCS block	10-15 m	3-5 days	1	None	None	~10 m^2 per sample
Geologic Coring	Extract sediment core	1-4 events (1-8 cores)	1 barge or ship, ~20 m	≥1/16 OCS block	<300 m	<3 days	1	None	None	~10 m^2 per sample
Grab Sampling	Collect sediment and benthic fauna	2-8 events (60-320 grabs)	1 barge or ship, ~20 m	≥1/16 OCS block	<1 m	<3 days	1	None	None	~10 m^2 per sample

Abbreviations: COST = Continental Offshore Stratigraphic Test; FAZ = Full Azimith Survey; N/A = Not applicable; OCS = Outer Continental Shelf; WAZ=Wide Azimith Survey.

[a] Shore base is the point of deployment to return berth.

[b] 1/16 of an OCS block (256 ac) is the smallest area considered for renewable energy leasing. All full-build out renewable energy projects in the Mid- and South Atlantic Planning Areas are wind park facilities that would be considerably larger than 1/16 of an OCS block. The average OCS wind park would be ≤10 OCS blocks in size.

[c] Would follow oil and gas program protocols for seismic airgun surveys.

Mitigation, Monitoring and Reporting Requirements

BOEM is proposing several measures to minimize potential adverse effects to listed species. These measures include limitations on airgun surveys in right whale critical habitat and their migratory corridors during certain times of the year, seismic, electromechanical and borehole operational protocols, vessel speed restrictions and marine trash and debris awareness briefings. Monitoring and reporting requirements are also proposed. A description of these measures and requirements follows.

Time-Area Considerations for North Atlantic Right Whales

BOEM is proposing to limit surveys using some active acoustic sound sources during critical times within and in proximity to right whale calving and nursing habitat, migratory pathways and when right whales are found aggregating in an area to limit acoustic and vessel traffic disturbance and collision risk for North Atlantic right whales. The Southeast seasonal management area (SMA) is a continuous area that extends 37 km from the shoreline from St. Augustine, Florida, to Brunswick, Georgia. Ten knot (18 km/hr) speed limits are in effect from November 15 to April 15, annually within the Southeast SMA. The Mid-Atlantic SMA is a combination of both continuous and 20 km arcs around the entrances to certain bays and ports with the same speed restrictions effective from November 1 through April 30, annually. Within the right whale critical habitat in the southeast the calving and nursing season extends from November 15 through April 15, annually (See Table 7). Dynamic management areas (DMAs) are designed to reduce the risk of whale-ship interactions when right whale(s) are found aggregating in an area.

Surveys using airguns would not be permitted within right whale critical habitat from November 15 through April 15 nor within southeast and mid-Atlantic SMAs from November 1 through April 30, nor within any DMAs that may be established by NMFS. Airgun use around these areas would include prohibition of sound levels entering these areas at 160 re 1 $\mu\text{Pa}\cdot\text{m}_{\text{rms}}$ or higher. See the section titled *Mitigation and Monitoring* for additional measures BOEM proposes for protection of right whales and special areas used by right whales.

BOEM also proposes additional mitigation and monitoring requirements for surveys using non-airgun HRG sources that are proposed for use within and adjacent to southeast critical habitat during right whale calving and nursing season and/or SMAs during restricted periods. Ten knot (18 km/hr) speed limits are in effect during applicable times and areas as listed above for airguns and in Table 7 below. Vessel operators would monitor the Early Warning System, Sighting Advisory System, and Mandatory Ship

Reporting System data to remain notified of right whale presence and be on alert as necessary. Active acoustic sources in use within a DMA must cease within 24 hours of the DMA's establishment. BOEM will review each of these non-airgun surveys to determine whether additional mitigation measures are necessary. See the section titled Mitigation and Monitoring for additional measures BOEM proposes for protection of right whales and special areas used by right whales.

Table 7: North Atlantic Right Whale Seasonal Management Areas and Critical Habitat within the Action Area and BOEM's Proposed Seismic and Oil and Gas Program HRG Restricted Activity Periods.

Individual Areas	Period of Restricted Activity	Restricted Area Latitude/Longitude
Entrance to Delaware Bay (Ports of Philadelphia and Wilmington)	Nov 1–April 30	Within a 20-nm (37 km) radius of the port entrance at 38°52′27.4″N/075°01′32.1″W
Entrance to Chesapeake Bay (Ports of Hampton Roads and Baltimore)		Within a 20-nm (37 km) radius of the port entrance at 37°00′36.9″N/75°57′50.5″W
Ports of Morehead City and Beaufort, NC		Within a 20-nm (37 km) radius of the port entrance at 34°41′32.0″N/076 °40′08.3″W
Wilmington, NC to Brunswick, GA		Within a continuous area 20 nm from shore between Wilmington, NC, to Brunswick, GA, bounded by the following Latitude/Longitude 34°10'30"N/77°49'12"W 33°56'42"N/77°31'30"W 33°36'30"N/77°47'06"W 33°28'24"N/78°32'30"W 32°59'06"N/78°50'18"W 31°50'00"N/80°33'12"W 31°27'00"N/80°51'36"W
Brunswick, GA to South of St. Augustine, FL		The area bounded by 31° 27′00.0″N/80°51′36.0″W and 29° 45′00.0″N/ 80°51′36.0″W
Southeast Right Whale Critical Habitat	Nov 15 – April 15	The area bounded by 31°15'N (approximately the mouth of the Altamaha River, GA) and 30°15'N (approximately Jacksonville, FL) from the shoreline out to 27 km (15 nmi) offshore, and the waters between 30°15'N and 28° 00 N (approximately Sebastian Inlet, FL) from the shoreline out to 9.26 km (5 nmi)

Right Whale Ship Strike Reduction Rule (50 CFR 224.105)

Seismic Airgun Survey Protocol

BOEM proposes to require certain protocols for all airgun surveys (deep penetration and airgun HRG surveys under the oil and gas program) including any surveys to evaluate formation suitability for carbon sequestration in the renewable energy program regardless of water depth. These protocols include exclusion zone determination, array ramp-up and shut down requirements and visual monitoring requirements by protected species observers prior to and during seismic airgun surveys (similar to BOEM and BSEE Joint NTL No. 2012-G02; BOEM and BSEE 2012a). These protocols apply to airguns only and not to electromechanical sources (side-scan sonars, boomers, sparkers, and chirp subbottom profilers, and single beam, swath or multibeam depth sounders) operating simultaneously with the airguns. The protocols also require the use of passive acoustic monitoring (PAM) to detect vocalizing marine mammals to allow operators to ramp-up and start/resume a seismic survey during times of reduced visibility (darkness, fog, rain, etc.) when such ramp-up otherwise would not be permitted using only visual observers.

Ramp-Up

Ramp-up (also known as "soft start") is conducted to increase the intensity of an airgun array over a period of 20 min or more, until maximum source levels are reached. Increasing sound levels are designed to warn animals of pending seismic operations, and to allow sufficient time for those animals to leave the immediate area and avoid the highest source levels from operating airguns. Proposed protocols for initiating ramp-up procedures during all seismic airgun survey operations, including airgun testing, are as follows. Only the procedures pertaining to ESA-listed species are shown here.

1. BOEM and BSEE will require operators to ramp-up airgun arrays. Protocols for initiating ramp-up procedures during all seismic airgun survey operations, including airgun testing, are as follows.
    a. Operators will visually monitor the exclusion zone and adjacent waters for the absence of marine mammals and sea turtles for at least 60 minutes before initiating ramp-up procedures. If no marine mammals or sea turtles are detected, ramp-up procedures may commence. Ramp-up at night or when the exclusion zone cannot be visually monitored is prohibited if the minimum source level drops below 160 dB re 1 μPa-m (rms) (see measure iv).
        i. Initiate ramp-up procedures by firing a single airgun. The preferred airgun to begin with should be the smallest airgun, in terms of energy output (dB) and volume (in.^{3}).
        ii. Continue ramp-up by gradually activating additional airguns over a period of at least 20 min, but no longer than 40 min, until the desired operating level of the airgun array is obtained.
        iii. Immediately shut down all airguns, ceasing seismic operations at any time a listed marine mammal or sea turtle is detected entering or within the exclusion zone. After a shutdown, ramp-up and seismic operations may recommence only when the exclusion zone has been visually inspected for at least 60 minutes to ensure the absence of marine mammals and sea turtles.

iv. Array source levels may be reduced using the same shot interval as the seismic survey, to maintain a minimum source level of 160 dB re 1 μPa-m (rms) for the duration of certain activities. The 60-minute visual clearance of the exclusion zone before ramp-up to full power is not required if the minimum source level of 160 dB re 1 μPa-m (rms) is maintained. Activities that are appropriate for maintaining the minimum source level are (1) all turns between transect lines, when a survey using the full array is being conducted immediately prior to the turn and will be resumed immediately after the turn; and (2) unscheduled, unavoidable maintenance of the airgun array that requires the interruption of a survey to shut down the array. The survey should be resumed immediately after the repairs are completed, but should not exceed 20 minutes. Use of the minimum source level to avoid the 60-minute visual clearance of the exclusion zone is only for events that occur during a survey using the full power array. The minimum sound source level is not to be used to allow a later ramp-up after dark or in conditions when ramp-up would not otherwise be allowed.

Exclusion Zone

The proposed seismic airgun survey protocol includes an exclusion zone to prevent exposure of marine mammals and sea turtles to the highest source levels from operating airguns. An exclusion zone is the area at and below the sea surface within a radius to be determined by calculating the maximum range at which animals could be exposed to a received SPL of 180 dB re 1 μPa, one of NMFS' criteria for Level A harassment (potential for injury) of cetaceans under the MMPA. The 180 dB re 1 μPa distance is calculated from the center of an airgun array. The radius of the exclusion zone must be calculated independently for each survey based on the configuration of the airgun array and the ambient acoustic environment, but must not be less than 500 m 1640 ft). This exclusion zone applies specifically to airguns, not electromechanical sources such as side-scan sonars, boomers, sparkers and chirp subbottom-profilers, and single beam or multibeam depth sounders that may be operating concurrently during seismic airgun surveys.

Visual Monitoring Methods for Seismic Airgun Surveys

The proposed seismic airgun survey protocol includes visual monitoring of the exclusion zone by trained protected species observers. All observers on duty will look for marine mammals and sea turtles using the naked eye and reticled, hand-held binoculars provided by the seismic vessel operator. The observers will stand watch in a suitable location that will not interfere with navigation or operation of the vessel and that affords the observers an optimal view of the sea surface. Observers will adjust their positions to have a 360^{o} view of the entire area around the seismic vessel, where possible, but ensure complete coverage of the exclusion zone. BOEM has proposed the following visual monitoring requirements:

1. Visual monitoring will begin no less than 60 minutes prior to the beginning of airgun ramp-up and continue until operations cease or sighting conditions do not allow observation of the sea surface (e.g., fog, rain, darkness).
2. If an ESA-listed marine mammal or sea turtle is observed, the observer should note and monitor the position (including latitude/longitude of the vessel and relative bearing and estimated distance to the animal) until the animal dives or moves out of visual range of the observer. Continue to observe for additional animals that may surface in the area, as often there are numerous animals that may surface at varying time intervals.
3. At any time an ESA-listed marine mammal or sea turtle is observed within the exclusion zone, whether due to the animal’s movement, the vessel’s movement, or because the animal surfaced inside the exclusion zone, the observer will call for the immediate shutdown of the operation, including airgun firing (the vessel may continue on its course but all airgun discharges must cease). The vessel operator must comply immediately with such a call by an on-watch visual observer. Any disagreement or discussion should occur only after shutdown.
4. After a shutdown, when no ESA-listed marine mammals or sea turtles are sighted for at least a 60-minute period, airgun ramp-up of the source array may begin. Ramp-up cannot begin unless conditions allow the sea surface to be visually inspected for ESA-listed marine mammals and sea turtles for 60 minutes prior to commencement of ramp-up. Ramp-up cannot begin after dark or in conditions that prohibit visual inspection (e.g., fog, rain, etc.) of the exclusion zone unless PAM is utilized.
5. Any shutdown due to an ESA-listed marine mammal or sea turtle sighting within the exclusion zone must be followed by a 60-minute all-clear period and then a standard, full ramp-up. Any shutdown for other reasons, including, but not limited to, mechanical or electronic failure, resulting in the cessation of the sound source for a period greater than 20 minutes, must also be followed by full ramp-up procedures.

6. Periods of airgun silence not exceeding 20 minutes in duration will not require ramp-up for the resumption of seismic operations if (a) visual surveys are continued diligently throughout the silent period (requiring daylight and reasonable sighting conditions), and (b) no ESA-listed marine mammals or sea turtles are observed in the exclusion zone. If ESA-listed marine mammals or sea turtles are observed in the exclusion zone during the short silent period, resumption of seismic survey operations must be preceded by ramp-up only after no ESA-listed marine mammals or turtles have been observed in the exclusion zone for a period of 60 minutes.

Shutdown Requirements

The proposed seismic airgun survey protocol will require shutdown of the airgun array any time a listed marine mammal or sea turtle is observed within the exclusion zone, whether due to the animal's movement, the vessel's movement, or because the animal surfaced inside the exclusion zone. In the event of a shutdown, seismic operations and ramp-up of airguns would resume only when the sighted animal has cleared the exclusion zone and no other marine mammals or sea turtles have been sighted within the exclusion zone for at least 60 minutes.

Non-Airgun HRG Survey Protocol for Renewable Energy and Marine Minerals Sites

The HRG surveys for renewable energy and marine minerals sites would use only electromechanical sources such as side-scan sonar, boomer and chirp subbottom profilers, and single beam, swath and multibeam depth sounders. BOEM proposes that all of their authorizations for non-airgun HRG surveys include requirements for visual monitoring of an exclusion zone by protected species observers and start-up and shut-down requirements as described below. The HRG surveys for oil and gas exploration typically use electromechanical sources operating concurrently with airgun arrays. These surveys would be subject to the proposed seismic airgun survey protocol described above.

Non-Airgun HRG Protocol Requirements

1. All HRG surveys must comply with requirements for vessel strike avoidance (See Vessel Strike Avoidance below). The recommended separation distance for North Atlantic right whales of 500 m (1,640 ft) would remain in effect during HRG surveys since it exceeds the exclusion zone radius specified below. Recommended separation distances for other whales and small cetaceans are less than 500 m, and would be superseded by the exclusion zone radius.

2. Sound sources used within Northern right whale critical habitat in the southeast during calving and nursing season should operate at frequencies above 30 kHz. If surveys are identified that require the use of frequencies below 30 kHz, BOEM should notify NMFS to determine if additional monitoring and mitigation measures are necessary.

3. For non-airgun HRG surveys within or adjacent to North Atlantic right whale critical habitat in the southeastern United States, within or adjacent to SMAs or within established DMAs, BOEM and BSEE must require operators implement the following measures :
    a. Vessel Strike Avoidance: A distance of 500 m (1, 1,640 ft) from right whales must be maintained during transit and operation and the Early Warning System, Sighting Advisory System, and Mandatory Ship Reporting System data notifying mariners of right whale presence must be monitored.
    b. A 200-m (656 ft) radius exclusion zone would be monitored by a protected species observer that would encompass the 180 dB re 1 μPa·m$_{rms}$ isopleth of any acoustic source. If the source produces a wider radius than this, than the exclusion zone would be increased to the field verification or modeled distance.
    c. Non-airgun HRG transmissions within a DMA must cease within 24 hours of the DMA's establishment.
4. At least one protected species observer would be required on watch aboard HRG survey vessels at all times during daylight hours (dawn to dusk – i.e., from about 30 minutes before sunrise to 30 minutes after sunset) when survey operations are being conducted, unless conditions (fog, rain, darkness) make sea surface observations impossible. If conditions deteriorate during daylight hours such that the sea surface observations are halted, visual observations must resume as soon as conditions permit. Ongoing activities may continue but may not be initiated under such conditions (i.e., without appropriate pre-activity monitoring).
5. Visual monitoring will begin no less than 60 minutes prior to start-up and continue until operations cease or sighting conditions do not allow observation of the sea surface (e.g., fog, rain, darkness).
6. At any time a marine mammal or sea turtle is observed within the exclusion zone, whether due to the animal's movement, the vessel's movement, or because the animal surfaced inside the exclusion zone, the observer will call for the immediate shutdown of the operation. The vessel operator must comply immediately with such a call by an on-watch visual observer. Any disagreement or discussion should occur only after shutdown.

The following additional requirements apply only to non-airgun HRG surveys in which one or more active acoustic sound sources will be operating at frequencies less than 200 kHz.

1. A 200-m (656-ft) radius exclusion zone will be established and monitored around the sound source. However, if this exclusion zone does not encompass the 180-dB radius, as validated through field verification or modeling, for any acoustic source proposed for use, BOEM may authorize, with NMFS' concurrence, surveys having a larger exclusion zone to encompass the 180-dB radius. Effectiveness can be evidenced by monitoring studies that have been carried out (i.e., Barkaszi et al 2012). Demonstration may also be in the class of boat used for the survey work, some of which may not have flying bridges or an observer height on deck that easily allows visual observation of the larger exclusion zone.
2. Active acoustic sound sources must not be activated until the protected species observer has reported the exclusion zone clear of all ESA-listed marine mammals and sea turtles for 60 minutes.
3. If any ESA-listed marine mammal or sea turtle is sighted at or within the exclusion zone, operators are required to immediately shutdown the equipment. Subsequent restart of the equipment may only occur following a confirmation that the exclusion zone if clear of ESA-listed marine mammals and sea turtles for 60 minutes.

Borehole Seismic Surveys

Borehole seismic surveys (completed during VSP surveys) differ from surface seismic surveys in a number of ways, including the use of much smaller airgun arrays, having an average survey time of 12-24 hrs, utilizing a sound source that may or may not move at 7.4-9.3 km/hr (4-5 kn), and requiring the capability of moving the receiver in the borehole between shots. Due to these differences, the following altered mitigation measures apply only to borehole seismic surveys:

1. During daylight hours, when visual observations of the exclusion zone are being performed as required in this protocol, borehole seismic operations will not be required to ramp-up for shutdowns of 30 min or less in duration, as long as no marine mammals or sea turtles are observed in the exclusion zone during the shutdown. If a marine mammal or sea turtle is sighted in the exclusion zone, ramp-up is required and may begin only after visual surveys confirm that the exclusion zone has been clear for 60 minutes.

2. Nighttime or poor visibility ramp-up is allowed only when passive acoustics are used to ensure that no marine mammals are present in the exclusion zone (as for all other seismic surveys). Operators are strongly encouraged to acquire the survey in daylight hours when possible.
3. Protected species observers must be used during daylight hours, as required in this protocol, and may be stationed either on the source boat or on the associated drilling rig or platform if a clear view of the exclusion zone and adjacent waters is available.
4. All other mitigations and provisions for seismic surveys as set forth in this protocol will apply to borehole seismic surveys.

Passive Acoustic Monitoring for Airgun Seismic Surveys

BOEM proposes to require operators to include passive acoustic monitoring (PAM) as part of the protected species observer program for seismic surveys. Operators using passive acoustic array devices are required to employ an observer proficient in passive acoustic monitoring operations and methods to detect vocalizing cetaceans. By using PAM devices, operators can ramp-up and start/resume a seismic survey during times of reduced visibility (e.g., darkness, fog, rain, etc.) when such ramp-up otherwise would not be permitted using only visual observers. All other mitigation measures required for seismic surveys remain applicable to seismic survey operations. A description of the passive acoustic system, the software used, and the monitoring plan should also be reported to BSEE prior to its use. NMFS will work with operators to determine which PAM systems are adequate for use.

Protected Species Observer Training for Seismic Airgun Surveys

BOEM and BSEE propose to require all visual observers complete a protected species observer training course that includes the requirements below. Operators may utilize observers trained by third parties based on the requirements below. All protected species observer training programs must:

BOEM and BSEE propose to require all visual observers to qualify for, and complete a protected species observer training course, as per the recommendations described in NOAA Fisheries Service 2012 National Standards for a Protected Species Observer and Data Management Program: A model for Seismic Surveys (Baker et al, unpublished) that includes the requirements below. All protected species observer training programs must:

1. Furnish BSEE a course information packet that includes the name and qualifications (i.e., experience, training completed, or educational background) of the instructor(s), the course outline or syllabus, and course reference material;

2. Furnish each trainee with a document stating successful completion of the course; and
3. Provide BSEE with names, affiliations, course completion dates for trainees.

The training course must include the following topics:

- i. Brief overview of the MMPA and the ESA as they relate to seismic acquisition and protection of marine mammals and sea turtles in the Atlantic Ocean.
- ii. Brief overview of seismic acquisition operations.
- iii. Overview of seismic mitigation measures and the protected species observer program.
- iv. Discussion of the role and responsibilities of the protected species observer, including:
  - a. Legal requirements (why you are here and what you do);
  - b. Professional behavior (code of conduct);
  - a. Integrity;
  - b. Authority of protected species observer to call for shutdown of seismic acquisition operations;
  - c. Assigned duties;
  - d. What can be asked of the observer;
    1. What cannot be asked of the observer; and
    2. Reporting of violations and coercion;
       3. Identification of Atlantic marine mammals and sea turtles;
       4. Cues and search methods for locating marine mammals and sea turtles; and,
       5. Distance determination techniques and training.
  - e. Data collection and reporting requirements:
       1. Forms and reports to BSEE via email on the 1st and 15th of each month; and,
       2. Marine mammal or sea turtle in exclusion zone/shutdown report within 24 hr.

Reporting Requirements for Seismic Airgun Surveys

BOEM and BSEE propose to require operators to submit observer effort reports and survey reports for each seismic and HRG survey conducted. In order to accommodate various vessels' bridge practices and preferences, vessel operators and observers may design data reporting forms in whatever format they deem convenient and appropriate. Alternatively, observers or vessel operators may adopt the United Kingdom's JNCC forms (available at their website, http://www.jncc.gov.uk). At a minimum, the following items should be recorded and included in reports to the BOEM and BSEE (see NOAA Fisheries Service 2012 National Standards for a Protected Species Observer and Data Management Program: A model for Seismic Surveys (Baker et al, unpublished).

Observer Effort Report: BOEM requires the submission of observer effort reports to BSEE on the 1st and the 15th of each month for each day seismic acquisition operations are conducted. These reports must include:

1. Vessel name;
2. Observers' names and affiliations;
3. Survey type (e.g., site, 3D, 4D);
4. BOEM Permit Number (for "off-lease seismic surveys") or OCS Lease Number (for "on-lease seismic surveys");
5. Date;
6. Time and latitude/longitude when daily visual survey began;
7. Time and latitude/longitude when daily visual survey ended; and
8. Average environmental conditions while on each visual survey rotation and session as well as when any conditions change during the rotation ,each session, including:
    a. Wind speed and direction;
    b. Sea state (glassy, slight, choppy, rough, or Beaufort scale);
    c. Swell (low, medium, high, or swell height in meters); and
    d. Overall visibility (poor, moderate, good).

Survey Report: BOEM requires the submission of survey reports to BSEE on the 1st and the 15th of the month for each day seismic acquisition operations are conducted and airguns are discharged. These reports must include:

1. Vessel name;
2. Survey type (e.g., site, 3D, 4D);
3. BOEM Permit Number (for "off-lease seismic surveys") or OCS Lease Number (for "on-lease seismic surveys"), if applicable;
4. Date;

5. Time pre-ramp-up survey begins;
6. Observations of marine mammals and sea turtles seen during pre-ramp-up surveys
7. Time ramp-up begins;
8. Observations of marine mammals and sea turtles seen during ramp-up;
9. Time sound source (airguns or HRG equipment) is operating at the desired intensity;
10. Observations of marine mammals and sea turtles seen during surveys;
11. If marine mammals or sea turtles were seen, was any action taken (i.e., survey delayed, guns shut down)?
12. Reason that marine mammals and sea turtles might not have been observed (e.g., swell, glare, fog); and
13. Time sound source (airgun array or HRG equipment) stops firing.

Sighting Report: BOEM shall require the submission of reports to BSEE for marine mammals and sea turtles sighted during seismic and HRG surveys on the 1st and the 15th of each month except as indicated below. These reports are in addition to any reports required as a condition of the geophysical permit and must include:

1. Vessel name;
2. Survey type (e.g., site, 3D, 4D);
3. BOEM Permit Number (for “off-lease seismic surveys”) or OCS Lease Number (for “on-lease seismic surveys”);
4. Date;
5. Time;
6. Watch status (Were you on watch or was this sighting made opportunistically by you or someone else?);
7. Observer or person who made the sighting;
8. Latitude/longitude of vessel;
9. Bearing of vessel; (true compass direction);
10. Bearing (true compass direction) and estimated range to animal(s) at first sighting;
11. Water depth (meters);
12. Species (or identification to lowest possible taxonomic level);
13. Certainty of identification (sure, most likely, best guess);
14. Total number of animals;
15. Number of juveniles;

16. Description (as many distinguishing features as possible of each individual seen, including length, shape, color and pattern, scars or marks, shape and size of dorsal fin, shape of head, and blow characteristics);
17. Direction of animal's travel – compass direction;
18. Direction of animal's travel – related to the vessel (drawing preferably);
19. Behavior (as explicit and detailed as possible; note any observed changes in behavior);
20. Activity of vessel;
21. Airguns firing? (yes or no); and
22. Closest distance (meters) to animals from center of airgun or airgun array (whether firing or not).

BOEM proposes to also require operators to report sightings of marine mammals or sea turtles within the exclusion zone that resulted in a shutdown of the airguns within 24 hours of the shut-down to BSEE. The sighting report must include the observed behavior of the animal(s) before shutdown, the observed behavior following shutdown (specifically noting any change in behavior), and the length of time between shutdown and subsequent ramp-up to resume the seismic survey (note if seismic survey was not resumed as soon as possible following shutdown). These sightings should also be included in the first regular semi-monthly report following the incident. Sighting forms should be scanned (or data typed) and sent via email to the BOEM and BSEE. If passive acoustic monitoring is used, an assessment must be included of the usefulness, effectiveness, and problems encountered with the use of that method of marine mammal detection in the reports described in this protocol.

Non Airgun HRG and Geotechnical Survey Reporting Requirements

Protected Species Observer Reports: Data on all protected-species observations must be recorded based on standard marine mammal observer collection data by the protected-species observer. This information must include:

1. Vessel name;
2. Observers' names, affiliations, and any Protected Species Training courses taken;
3. Date;
4. Time and latitude/longitude when daily visual survey began;
5. Time and latitude/longitude when daily visual survey ended; and
6. Average environmental conditions during visual surveys including:
    a. Wind speed and direction;

   b. Sea state (glassy, slight, choppy, rough, or Beaufort scale);
   c. Swell (low, medium, high, or swell height in meters); and
   d. Overall visibility (poor, moderate, good).

7. Species (or identification to lowest possible taxonomic level);
8. Certainty of identification (sure, most likely, best guess);
9. Total number of animals;
10. Number of juveniles;
11. Description (as many distinguishing features as possible of each individual seen, including length, shape, color and pattern, scars or marks, shape and size of dorsal fin, shape of head, and blow characteristics);
12. Direction of animal's travel – related to the vessel (drawing preferably);
13. Behavior (as explicit and detailed as possible; note any observed changes in behavior);
14. Activity of vessel when sighting occurred.

Non-Airgun and Geotechnical Reporting Requirements: The lessee/operator must provide BOEM with a report within ninety (90) calendar days following the commencement of HRG and/or geotechnical sampling activities that includes all protected species observer reports, a summary of the survey activities and an estimate of the number of listed marine mammals and sea turtles observed or taken during these survey activities.

Guidance for Vessel Strike Avoidance

All authorizations for shipboard surveys would include guidance for protected species identification, vessel strike avoidance and injured/dead protected species reporting. The guidance would be similar to Joint BOEM-BSEE NTL 2012-G01 (*Vessel Strike Avoidance and Injured/Dead Protected Species Reporting*) (USDOI, BOEM and BSEE, 2012b). The lessee and/or operator must ensure that all vessels conducting G&G activities comply with the vessel strike avoidance measures specified below except under extraordinary circumstances when the safety of the vessel or crew are in doubt or the safety of life at sea is in question. The vessel strike avoidance measures have been included in the proposed siesmic airgun and non-airgun HRG survey protocols and include maintaining a vigilant watch for listed whales and sea turtles and slowing down or stopping vessels to avoid striking protected species, observing the 10 kt (18.5 km/h) speed restriction in SMAs, North Atlantic right whale southeast critical habitat and within DMAs, maintaining separation distances from listed whales and sea turtles and reporting of any dead or injured listed whales or sea turtles.

Guidance for Marine Debris Awareness

All authorizations for shipboard surveys would include guidance for marine debris awareness. The deliberate discharge of containers and other similar materials (i.e., trash and debris) into the marine environment is generally prohibited unless it is passed through a comminutor that breaks up solids and can pass through a 25-mm mesh screen. Discharge of plastic is prohibited regardless of size. Durable identification marking on equipment, tools and containers (especially drums), and other material are also required as well as recording and reporting of items lost overboard. Special precautions should be taken when handling and disposing of small items and packaging materials, particularly those made of non-biodegradable, environmentally persistent materials such as plastic or glass that can be lost in the marine environment and washed ashore.

All vessel operators, employees and contractors actively engaged in G&G surveys must be briefed on marine trash and debris awareness elimination as described in BSEE NTL No. 2012-G01 (*Marine Trash and Debris Awareness and Elimination*) (USDOI, BSEE, 2012a). BOEM will not require G&G operators, employees and contractors to undergo formal training or to post placards. The operator will be required to ensure that its employees and contractors are made aware of the environmental and socioeconomic impacts associated with marine trash and debris and their responsibilities for ensuring that trash and debris are not intentionally or accidentally discharged into the marine environment.

Overview of NMFS' Assessment Framework

We approach the assessments in section 7 consultations using several steps. The first step identifies a proposed action and those aspects of an action that are likely to have direct or indirect effects on the physical, chemical, and biotic environment of an action area. This step includes identifying the specific stressors produced by the action, including defining the spatial extent of these stressors and changes in that spatial extent over time (the spatial extent of these stressors is the "Action Area" for a consultation).

The second step starts by identifying the threatened species, endangered species, and designated critical habitat (hereafter, "listed resources") that are likely to co-occur with these effects in space and time and the nature of that co-occurrence (the latter represent our exposure analyses). We try to identify the number, age (or life stage), and gender of individuals that are likely to be exposed to an Action's effects and the populations or subpopulations those individuals represent.

Once we identify which listed resources are likely to be exposed to an Action's effects and the nature of that exposure, we evaluate the available evidence to determine how those listed resources are likely to respond given their probable exposure (these represent our response analyses). The final steps of our assessments are different for listed species and designated critical habitat. For listed species, we assess the consequences of these responses to the individuals that have been exposed, the populations those individuals represent, and the species those populations comprise. For designated critical habitat, we assess the consequences of these responses on the value of the critical habitat for the conservation of the species for which the habitat had been designated.

Application of the General Approach to this Consultation

The primary stressors associated with the G&G activities BOEM proposes to authorize within the Action Area consist of:

1. sound fields produced by active seismic and electromechanical sources used during G&G activities (airguns, boomers, sparkers, side-scan sonars, depth sounders and subbottom profilers);
2. disturbance produced by vessels and equipment involved in G&G activities;
3. the risk of collisions associated with vessels involved in G&G activities;
4. disturbance produced by aircraft traffic;
5. drilling discharges and accidental fuel spills;
6. trash and debris; and,
7. seafloor disturbance from coring, equipment or anchors associated with certain deep penetration surveys, drilling and sampling activities.

The first step of our analysis evaluates the available evidence to determine the likelihood of listed species or critical habitat being exposed to these potential stressors. Our analysis assumes that these stressors pose no risk to listed species or critical habitat if these potential stressors do not co-occur, in space or time, with (1) individuals of endangered or threatened species or their designated critical habitat; (2) species that are food for endangered or threatened species; (3) species that prey on or compete with endangered or threatened species; or (4) pathogens for endangered or threatened species.

Exposure Analyses

As discussed in the introduction to this section, our exposure analyses are designed to identify the listed resources that are likely to co-occur with these effects in space and time and the nature of that co-occurrence. Our exposure analyses are designed 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. When it is impossible or impracticable to estimate the number of individuals likely to be exposed, we try to estimate the proportion of a population that is likely to be exposed. If we cannot estimate this proportion, we will rely on a surrogate or index.

For our exposure analyses, NMFS generally relies on an action agency's estimates of the number of marine mammals that might be "taken" (as that term is defined for the purposes of the MMPA). In a small number of consultations involving active acoustic sound sources, however, NMFS conducted separate analyses to estimate the number of endangered or threatened marine animals that might be exposed to stressors produced by a proposed action to assess the effect of assumptions in an action agency's model estimates. For example, NMFS' model was used to independently estimate the number of marine mammals that might be exposed to U.S. Navy training activities in a few recent consultations. These consultations identified the sole or primary stressor as an active sound source (mid-frequency sonar) and data were available on (1) densities of endangered or threatened animals in an action area, (2) the ship speeds, (3) the radial distance at which different received levels would be detected from a source given sound speed profiles levels (estimated sound propagation from the highest received levels close to the source with increasing distance from the source down to ambient sound levels), and (4) the duration of specific training exercises in predictable areas of the associated training range.

Although for this consultation data were available for densities of marine mammals and ship speeds, we did not have the radial distance at which different received levels would be detected with increasing distance from a source down to ambient sound levels in the Action Area given sound speed profiles. BOEM only modeled radial distances for received levels of sound between 210 dB and 150 dB re 1 μPa, for their representative airgun array and single airgun, boomer, side scan sonar, chirp subbottom profiler and multibeam depth sounder, and therefore only those radial distances were available to us. Also, individual surveys and simultaneous, overlapping and non-overlapping survey activity can occur anywhere at any time within or adjacent to the Mid- and South Atlantic Planning Areas and it is impossible to predict the times, areas and durations of the different types of seismic surveys and HRG surveys that would be representative of the spatial and temporal arrangements of the surveys that would occur in any particular year, let alone over the life time of the proposed action. For the two reasons just cited, we could not run the exposure models we had developed as described for other consultations.

BOEM's estimate of the number of animals that might be "taken" follows two steps: estimating the acoustic field produced from a sound source and how it propagates through the water; and determining the net exposure of marine mammals that occur within the ensonified volume of water, using the MMPA "take" criteria recommended by NMFS. BOEM's model starts with a set of representative acoustic modeling regions based on seasonal sound velocity profiles, bathymetries and bottom types within the Action Area for this consultation. Within each of these regions, transmission loss was modeled for representative sound sources (airguns, side-scan sonar, boomer, chirp subbottom profiler and multibeam depth sounder) at depth which were used to estimate the acoustic field radiated by the sound sources. BOEM also estimated sound exposure levels and root mean square (rms) levels.

BOEM estimated the expected number of animals that might be exposed to peak pressures based on levels that exceed certain regulatory thresholds using the acoustic fields estimated for the representative sound sources and the Acoustic Integration Model (AIM). AIM predicts exposure of receivers (in this case, marine species) to any stimulus propagating through space and time.

We use BOEM's estimates as a rough estimate of the number of instances in which endangered and threatened species might be exposed to and "taken" by sounds produced by seismic and HRG survey activities along the Atlantic seaboard. However, these "take" estimates underestimate the number of instances in which these species would have been exposed to those sounds and would have responded in ways that we would consider "adverse", even if those adverse responses did not constitute one or more forms of "take." Nevertheless, we relied solely on the results of acoustic models and the marine mammal take model BOEM used to prepare its NEPA compliance documents.

Response Analyses

As discussed in the introduction to this section, once we identify which listed resources are likely to be exposed to active seismic and electromechanical sources associated with the proposed G&G activities and the nature of that exposure, we examined the scientific and commercial data available to determine whether and how (1) endangered or threatened species are likely to respond following exposure and the set of physical, physiological, behavioral, or social responses that are likely and (2) the quantity, quality, or availability of one or more of the physical or biological features that led us to conclude that the area was essential for the conservation of a particular listed species are likely to change in response to the exposure.

Conceptual Model for Response Analyses

To guide our response analyses, we constructed a conceptual model that is based on a model of animal behavior and behavioral decision-making and incorporates the cognitive processes involved in behavioral decisions (Figure 3) although we continue to recognize the risks presented by physical trauma and noise-induced losses in hearing sensitivity (threshold shift). This model is also based on a conception of "hearing" that includes cognitive processing of auditory cues, rather than focusing solely on the mechanical processes of the ear and auditory nerve. Our model incorporates the primary mechanisms by which behavioral responses affect the longevity and reproductive success of animals: changing an animal's energy budget, changing an animal's time budget (which is related to changes in an animal's energy budget), forcing animals to make life history trade-offs (for example, engaging in evasive behavior such as deep dives that involve short-term risks while promoting long-term survival), or changes in social interactions among groups of animals (for example, interactions between a cow and her calf).

This conceptual model begins with the specific acoustic stimuli that we focus on in an assessment (Box 1 in Figure 3). Although we generally considered different acoustic stimuli separately, we considered a single source of multiple acoustic stimuli as a complex "acoustic object" that has several acoustic properties. For example, we treat pulses produced by active sound sources and sounds produced by the source vessel as a single "acoustic object" that produced continuous sounds (engine-noise, propeller cavitation, hull displacement, etc.) and periodic impulsive pulses. Because animals would be exposed to this complex of sounds produced by a single, albeit moving, source over time, we assumed they would generally respond to the acoustic stream associated with this single acoustic object moving through their environment. Multiple ships associated with a particular type of survey, for instance WAZ surveys, are expected to also represent a single acoustic object as all vessels are moving in formation at the same speeds while alternating shots. Multiple ships associated with other surveys, such as support ships for different types of surveys that move independently of the survey formation would represent different acoustic objects in the acoustic scene of endangered and threatened marine animals.

Acoustic stimuli can represent two different kinds of stressors: *processive stressors*, which require high-level cognitive processing of sensory information, and *systemic stressors*, which usually elicit direct physical or physiological responses and, therefore, do not require high-level cognitive processing of sensory information (Anisman and Merali 1999, de Kloet *et al*. 2005, Herman and Cullinan 1997). Disturbance from surface vessels and airguns would be examples of processive stressors while ship strikes would be an example of a systemic stressor. As a result, G&G activities may result in two general classes of responses:

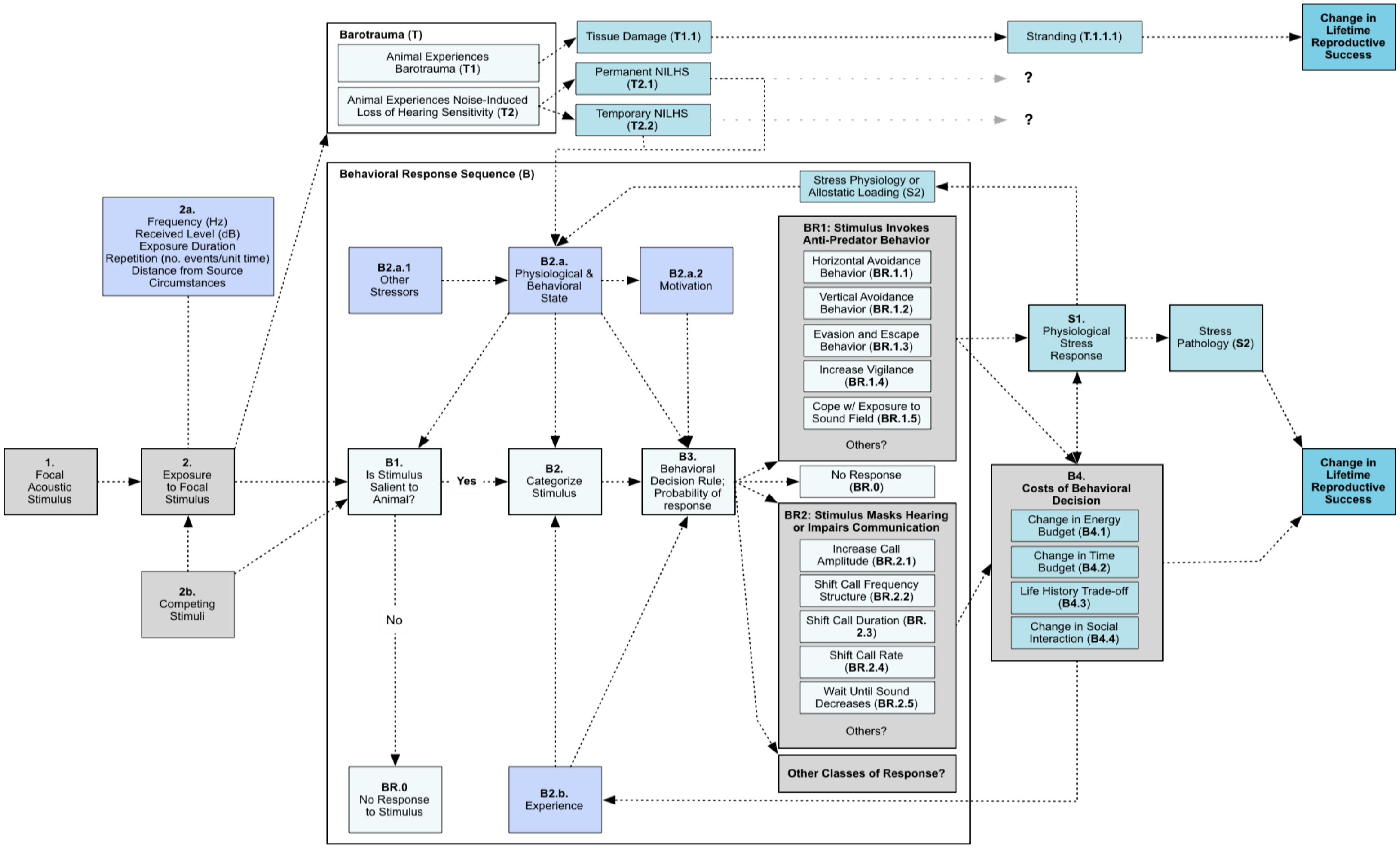
1. responses that are influenced by an animal's assessment of whether a potential stressor poses a threat or risk (see Figure 3: Behavioral Response).
2. responses that are not influenced by the animal's assessment of whether a potential stressor poses a threat or risk (see Figure 3: Physical Damage).

Our conceptual model explicitly recognizes that other acoustic and non-acoustic stimuli that occur in an animal's environment might determine whether a focal stimulus is salient to a focal animal (the line connecting Box 2b to Box 2 in Figure 3). The salience of an acoustic signal will depend, in part, on its signal-to-noise ratio and, given that signal-to-noise ratio, whether an animal will devote attentional resources to the signal or other acoustic stimuli (or ambient sounds) that might compete for the animal's attention (the line connecting Box 2b to Box B1 in Figure 3)[1]. That is, an acoustic signal might not be salient (1) because of a signal-to-noise ratio or (2) because an animal does not devote attentional resources to the signal, despite its signal-to-noise ratio. Absent information to the contrary, we generally assume that an acoustic stimulus that is "close" to an animal (within 10 – 15 kilometers) would remain salient regardless of competing stimuli and would compete for an animal's attentional resources. By extension, we also assume that any behavioral change we might observe in an animal would have been caused by a focal stimulus (the stimulus most immediately confronting the animal) rather than competing stimuli. However, as the distance between the source of a specific acoustic signal and a receiving animal increases, we assume that the receiving animal is less likely to devote attentional resources to the signal.

If we conclude (or if we assume) that an acoustic stimulus, such as a seismic or electromechanical source, was salient to an animal or population of animals, we would then ask how an animal might classify the stimulus as a cue about its environment (Box B2 in Figure 3) because an animal's response to a stimulus in its environment depends upon whether and how the animal converts the stimulus into information about its environment (Blumstein and Bouskila 1996, Yost 2007). For example, if an animal classifies a stimulus as a "predatory cue," that classification will invoke a suite of candidate physical, physiological, or behavioral responses that are appropriate to being confronted by a predator (this would occur regardless of whether a predator is, in fact, present).

---

[1] See Blumstein and Bouskila (1996) for more of a review of the literature on how animals process and filter sensory information, which affects the subjective salience of sensory stimuli. See Clark and Dukas (2003), Dukas (1998, 2002, 2004), and Roitblat (1987) for more extensive reviews of the literature on attentional processes and the consequences of limited attentional resources in animals.



28 February 2012

Figure 3. Conceptual model of the potential responses of endangered and threatened species upon being exposed to an active seismic and electromechanical sources and the pathways by which those responses might affect the fitness of individual animals that have been exposed. See text in “Application of this Approach” and “Response Analyses” for an explanation of the model and supporting literature.

By incorporating a more expansive concept of “hearing,” our conceptual model departs from earlier models which have focused on the mechanical processes of “hearing” associated with structures in the ear that transduce sound pressure waves into vibrations and vibrations to electro-chemical impulses. That conception of hearing resulted in assessments that focus almost exclusively on active acoustic sources while discounting other acoustic stimuli associated with activities that marine animals might also perceive as relevant. That earlier conception of hearing also led to an almost singular focus on the intensity of the sound - its received level (in decibels) - as an assessment metric and noise-induced hearing loss as an assessment endpoint.

Among other considerations, the earlier focus on received level and losses in hearing sensitivity failed to recognize several other variables that affect how animals are likely to respond to acoustic stimuli:

1. “hearing” includes the cognitive processes an animal employs when it analyzes acoustic impulses (see Bregman 1990, Blumstein and Bouskila 1996, Hudspeth 1997, Yost 2007), which includes the processes animals employ to integrate and segregate sounds and auditory streams and the circumstances under which they are likely to devote attentional resources to an acoustic stimulus.
2. animals can “decide” which acoustic cues they will focus on and their decision will reflect the salience of a cue, its spectral qualities, and the animal’s physiological and behavioral state when exposed to the cue.
3. animals not only perceive the received level (in dB) of a sound source, they also perceive their distance from a sound source. Further, animals are more likely to devote attentional resources to sounds that are close than sounds that are distant.
4. both received levels and the spectral qualities of sounds degrade over distance so the sound perceived by a distant receiver is not the same sound at the source.

As a result of this shift in focus, we have to consider more than the received level of a particular low- or mid-frequency wave form and its effects on the sensitivity of an animal’s ear structure. We also have to distinguish between different auditory scenes; for example, animals will distinguish between sounds from a source that is moving away, sounds produced by a source that is approaching them, sounds from multiple sources that are all approaching, sounds from multiple sources that appear to be moving at random, etc.

Animals would then combine their perception of the acoustic stimulus with their assessment of the auditory scene (which include other acoustic stimuli), their awareness of their behavioral state, physiological state, reproductive condition, and social circumstances to assess whether the acoustic stimulus poses a risk and the degree of risk it might pose, whether it is impairing their ability to communicate with conspecifics, whether it is impairing their ability to detect predators or prey, etc. We assume that animals would categorize an acoustic source differently if the source is moving towards its current position (or projected position), moving away from its current position, moving tangential to its current position, if the source is stationary, or if there are multiple acoustic sources it its auditory field.

This process of "categorizing a stimulus" (Box B2 in Figure 3) lends meaning to a stimulus and places the animal in a position to decide whether and how to respond to the stimulus (Blumstein and Bouskila 1996). How an animal categorizes a stimulus will determine the set of candidate responses that are appropriate in the circumstances. That is, we assume that animals that categorizes a stimulus as a "predatory cue" would invoke candidate responses that consisted of anti-predator behavior rather than foraging behavior (Bejder *et al*. 2009, Blumstein and Bouskila 1996).

We then assume that animals apply one or more behavioral decision rules to the set of candidate responses that are appropriate to the acoustic stimulus as it has been classified (Box B3 in Figure 3). Our use of the term "behavioral decision rule" follows Blumstein and Bouskila (1996), Dill (1987), McFarland (1982), and Lima and Dill (1990) and is synonymous with the term "behavioral policy" of McNamara and Houston (1986): the process an animal applies to determine which specific behavior it will select from the set of behaviors that are appropriate to the auditory scene, given its physiological and behavioral state when exposed and its experience. Because we would never know the behavioral policy of an individual, free- ranging animal, we treat this policy as a probability distribution function that matches a particular response in the suite of candidate behavioral responses.

Once an animal selects a behavioral response from a set of candidate behaviors, we assume that any change in behavioral state would represent a shift from an optimal behavioral state (or behavioral act) to a sub-optimal behavioral state (or behavioral act) as the animal responds to a stimulus such as acoustic sound sources. That selection of the sub-optimal behavioral state or act could be accompanied by *canonical costs*, which are reductions in the animal's expected future reproductive success that would occur when an animal engages in suboptimal behavioral acts (McNamara and Houston 1986). Specifically, canonical costs represent a reduction in current and expected future reproductive success (which integrates survival and longevity with current and future reproductive success) that would occur when an animal engages in a sub-optimal rather than an optimal sequence of behavioral acts; given the pre-existing physiological state of the animal in a finite time interval (Barnard and Hurrst 1996, Houston 1993, McFarland and Sibly 1975, McNamara 1993, McNamara and Houston 1982, 1986, 1996; Nonacs 2001). Canonical costs would generally result from changes in animals' energy budgets (McEwen and Wingfield 2003, Moberg 2000; Romero 2004, Sapolsky 1990, 1997), time budgets (Frid and Dill 2002, Sutherland 1996), life history trade-offs (Cole 1954, Stearns 1992), changes in social interactions (Sutherland 1996), or combinations of these phenomena (see Box B4 in Figure 3). We assume that an animal would not incur a canonical cost if they adopted an optimal behavioral sequence (see McNamara and Houston 1986 for further treatment and discussion).

This conceptual model does not require us to assume that animals exist in pristine environments; in those circumstances in which animals are regularly or chronically confronted with stress regimes that animals would adapt to by engaging in sub-optimal behavior, we assume that a change in behavior that resulted from exposure to a particular stressor or stress regime would either contribute to sub-optimal behavior or would cause animals to engage in behavior that is even further from optimal.

Risk Analyses

As discussed in the Introduction to this section, the final steps of our analyses -- establishing the risks those responses pose to endangered and threatened species or designated critical habitat -- normally begin by identifying the probable risks actions pose to listed individuals that are likely to be exposed to an action's effects. Our analyses then integrate those individual's risks to identify consequences to the populations those individuals represent. Our analyses conclude by determining the consequences of those population-level risks to the species those populations comprise.

We measure risks to listed individuals using the concept of current or expected future reproductive success which, as we described in the preceding sub-section, integrate survival and longevity with current and future reproductive success. In particular, we examine the scientific and commercial data available to determine if an individual's probable response to stressors produced by an Action would reasonably be expected to reduce the individual's current or expected future reproductive success by increasing the individual's likelihood of dying prematurely, having reduced longevity, increasing the age at which individuals become reproductively mature, reducing the age at which individuals stop reproducing, reducing the number of live births an individual produces during any reproductive bout, reducing the number of times an individual is likely to reproduce over the reproductive lifespan (in animals that reproduce multiple times), or causing an individual's progeny to experience any of these phenomena.

When individual animals would be expected to experience reductions in their current or expected future reproductive success, we would also expect those reductions to also reduce the abundance, reproduction rates, or growth rates (or increase variance in one or more of these rates) of the populations those individuals represent (see Stearns 1992). If we conclude that listed animals are *not* likely to experience reductions in their current or expected future reproductive success, we would conclude our assessment.

If we conclude that listed animals are likely to experience reductions in their current or expected future reproductive success, we would integrate those individuals risks to determine if the number of individuals that experience reduced fitness (or the magnitude of any reductions) is likely to be sufficient to reduce the viability of the populations those individuals represent (measured using changes in the populations' abundance, reproduction, spatial structure and connectivity, growth rates, or variance in these measures to make inferences about a population's probability of becoming demographically, ecologically, or genetically extinct in 10, 25, 50, or 100 years). For this step of our analyses, we would rely on the population's base condition (established in the *Environmental Baseline* and *Status of Listed Resources* sections of this Opinion) as our point of reference.

Our risk analyses normally conclude by determining whether changes in the viability of one or more population is or is not likely to be sufficient to reduce the viability of the species (measured using probability of demographic, ecological, or genetic extinction in 10, 25, 50, or 100 years) those populations comprise. For these analyses, we combine our knowledge of the patterns that accompanied the decline, collapse, or extinction of populations and species that have experienced these phenomena in the past as well as population viability models, if available.

Our assessment is designed to establish that a decline, collapse, or extinction of an endangered or threatened species is not likely to occur; we do not conduct these analyses to establish that such an outcome is likely to occur. For this step of our analyses, we would also use the species' status (established in the *Status of the Species* section of this Opinion) as our point of reference.

For critical habitat our "destruction or adverse modification" determinations must be based on an action's effects on the conservation value of habitat that has been designated as critical to threatened or endangered species. If an area encompassed in a critical habitat designation is likely to be exposed to the direct or indirect consequences of the proposed action on the natural environment, we ask if primary constituent elements included in the designation (if there are any) or physical, chemical, or biotic phenomena that give the designated area value for the conservation are likely to respond to that exposure.

If primary constituent elements of designated critical habitat (or physical, chemical, or biotic phenomena that give the designated area value for the conservation of listed species) are likely to respond given exposure to the direct or indirect consequences of the proposed action on the natural environment, we ask if those responses are likely to be sufficient to reduce the quantity, quality, or availability of those constituent elements or physical, chemical, or biotic phenomena.

If the quantity, quality, or availability of the primary constituent elements of the area of designated critical habitat (or physical, chemical, or biotic phenomena) are reduced, we ask if those reductions are likely to be sufficient to reduce the conservation value of the designated critical habitat for listed species in the action area.

If the conservation value of designated critical habitat in an action area is reduced, the final step of our analyses asks if those reductions are likely to be sufficient to reduce the conservation value of the entire critical habitat designation. We use the conservation value of the entire designated critical habitat as our point of reference for this comparison. For example, if the designated critical habitat has limited current value or potential value for the conservation of listed species, the limited value is our point of reference for our assessment.

Evidence Available for the Consultation

To conduct these analyses, we considered lines of evidence available through published and unpublished sources that represent evidence of adverse consequences or the absence of such consequences. A considerable body of scientific information on anthropogenic sound and its effects on marine mammals and other marine life has become available. Many investigators have studied the potential responses of marine mammals and other marine organisms to human-generated sounds in marine environments or have integrated and synthesized the results of these studies for example, Abgrall *et al.* 2008, Bowles *et al.* 1994; Croll *et al.* 1999, 2001; Frankel and Clark 1998; Gisiner 1998, Gordon et al 2004; McCauley and Cato 2001; NRC 1994 1996, 2000, 2003, 2005; Norris 1994; OSPAR 2009; Reeves 1992; Richardson *et al.* 1995; Southall *et al*. 2007; Tyack 2000, 2007; Weilgart 2007; Wright *et al.* 2007; Tyack and Clark 2000; Popper and Hastings 2009; Normandeau Associates, Inc. 2012. To comply with our obligation to use the best scientific and commercial data available, we conducted additional searches to identify information relevant to the potential stressors and responses of marine mammals, sea turtles and fish species that may be affected by the proposed action to draw conclusions about the likely risks to the continued existence of these species. Despite the information that has become available, this assessment involves a large amount of uncertainty about the basic hearing capabilities of marine mammals; how marine mammals use sounds as environmental cues, how they perceive acoustic features of their environment; the importance of sound to the normal behavioral and social ecology of marine mammals; the mechanisms by which human-generated sounds affect the behavior and physiology (including the non-auditory physiology) of marine mammals, and the circumstances that are likely to produce outcomes that have adverse consequences for individual marine mammals and marine mammal populations. These same questions remain for sea turtles and fish species addressed in this consultation, however, there are even less studies available for these taxa.

Treatment of "Cumulative Impacts" (in the sense of NEPA)

Several organizations have argued that previous biological opinions failed to consider the "cumulative impact" (in the NEPA sense of the term) of anthropogenic sound on the ocean environment and its organisms, particularly endangered and threatened species and critical habitat that has been designated for them (for example, see NRDC 2007 and Ocean Mammal Institute 2007). In each instance, we have had to explain how biological opinions consider "cumulative impacts" (in the NEPA sense of the term).

The U.S. Council on Environmental Quality defined "cumulative impacts" (which we refer to as "cumulative effects" to distinguish between NEPA and ESA uses of the same term) as "the impact on the environment which results from the incremental impact of the action when added to other past, present, and reasonably foreseeable future actions regardless of what agency (Federal or non-federal) or person undertakes such other actions" (40 CFR 1508.7). The effects analyses of biological opinions considers the "impacts" on listed species and designated critical habitat that result from the incremental impact of an action by identifying natural and anthropogenic stressors that affect endangered and threatened species throughout their range (the *Status of the Species*) and within an Action Area (the *Environmental Baseline*, which articulate the pre-existing *impacts* of activities that occur in an Action Area, including the past, contemporaneous, and future *impacts* of those activities). We assess the effects of a proposed action by adding their direct and indirect effects to the *impacts* of the activities we identify in an *Environmental Baseline* (50 CFR 402.02), in light of the impacts of the status of the listed species and designated critical habitat throughout their range; as a result, the results of our effects analyses are similar to those contained in the "cumulative impact" sections of NEPA documents.

A Brief Background on Sound

Sound is a wave of pressure variations propagating through a medium (for this consultation, the sounds generated by seismic and electromechanical equipment proparates through marine water as its medium). Pressure variations are created by compressing and relaxing the medium. Sound measurements can be expressed in two forms: *intensity* and *pressure*. Acoustic intensity is the average rate of energy transmitted through a unit area in a specified direction and is expressed in watts per square meter (W/m^2). Acoustic intensity is rarely measured directly, it is derived from ratios of *pressures*; the standard reference pressure for underwater sound is 1 microPascal (μPa); for airborne sound, the standard reference pressure is 20 μPa (Richardson et al 1995a).

Acousticians have adopted a logarithmic scale for sound intensities, which is denoted in decibels (dB). Decibel measurements represent the ratio between a measured pressure value and a reference pressure value (in this case 1 μPa or, for airborne sound, 20 μPa.). The logarithmic nature of the scale means that each 10 dB increase is a ten-fold increase in power (e.g., 20 dB is a 100-fold increase, 30 dB is a 1,000-fold increase). The term "sound pressure level" implies a decibel measure and a reference pressure that is used as the denominator of the ratio. Throughout this Opinion, we use 1 microPascal (denoted re: 1μPa) as a standard reference pressure unless noted otherwise.

It is important to note that decibels underwater and decibels in air are not the same and cannot be directly compared. Because of the different densities of air and water and the different decibel standards in water and air, a sound with the same intensity (i.e., power) in air and in water would be approximately 63 dB quieter in air.

Sound frequency is measured in cycles per second, or Hertz (abbreviated Hz), and is analogous to musical pitch; high-pitched sounds contain high frequencies and low-pitched sounds contain low frequencies. Natural sounds in the ocean span a huge range of frequencies: from earthquake noise at 5 Hz to harbor porpoise clicks at 150,000 Hz. These sounds are so low or so high in pitch that humans cannot even hear them; acousticians call these infrasonic and ultrasonic sounds, respectively. A single sound may be made up of many different frequencies together. Sounds made up of only a small range of frequencies are called "narrowband", and sounds with a broad range of frequencies are called "broadband"; airguns are an example of a broadband sound source and sonars are an example of a narrowband sound source.

When considering the influence of various kinds of noise on the marine environment, it is necessary to understand that different kinds of marine life are sensitive to different frequencies of sound. Most dolphins, for instance, have excellent hearing at very high frequencies between 10,000 and 100,000 Hz. Their sensitivity at lower frequencies below 1000 Hz, however, is quite poor. On the other hand, the hearing sensitivity of most sea turtles appears to be best at frequencies between about 200 Hz and 700 Hz. As a result, sea turtles might be expected to suffer more harmful effects from low frequency noise than would dolphins.

When sound travels away from its source, its loudness decreases as the distance traveled by the sound increases. Thus, the loudness of a sound at its source is higher than the loudness of that same sound a kilometer distant. Acousticians often refer to the loudness of a sound at its source as the *source level* and the loudness of sound elsewhere as the *received level*. For example, a humpback whale 3 kilometers from an airgun that has a source level of 230 dB may only be exposed to sound that is 160 dB loud. As a result, it is important not to confuse source levels and received levels when discussing the loudness of sound in the ocean.

As sound moves away from a source, its propagation in water is influenced by various physical characteristics, including water temperature, depth, salinity, and surface and bottom properties that cause refraction, reflection, absorption, and scattering of sound waves. Oceans are not homogeneous and the contribution of each of these individual factors is extremely complex and interrelated. Sound speed in seawater is generally about 1,500 meters per second (5,000 feet per second) although this speed varies with water density, which is affected by water temperature, salinity (the amount of salt in the water), and depth (pressure). The speed of sound increases as temperature and depth (pressure), and to a lesser extent, salinity, increase. The variation of sound speed with depth of the water is generally presented by a “sound speed profile,” which varies with geographic latitude, season, and time of day.

Sound tends to follow many paths through the ocean, so that a listener may hear multiple, delayed copies of transmitted signals (Richardson et al 1995a). Echoes are a familiar example of this phenomenon in air. In order to determine what the paths of sound transmission are, one rule is to seek paths that deliver the sound to the receiver the fastest. If the speed of sound were constant throughout the ocean, acoustic rays would consist of straight-line segments, with reflections off the surface and the bottom. However, because the speed of sound varies in the ocean, most acoustic rays do not follow a straight path.

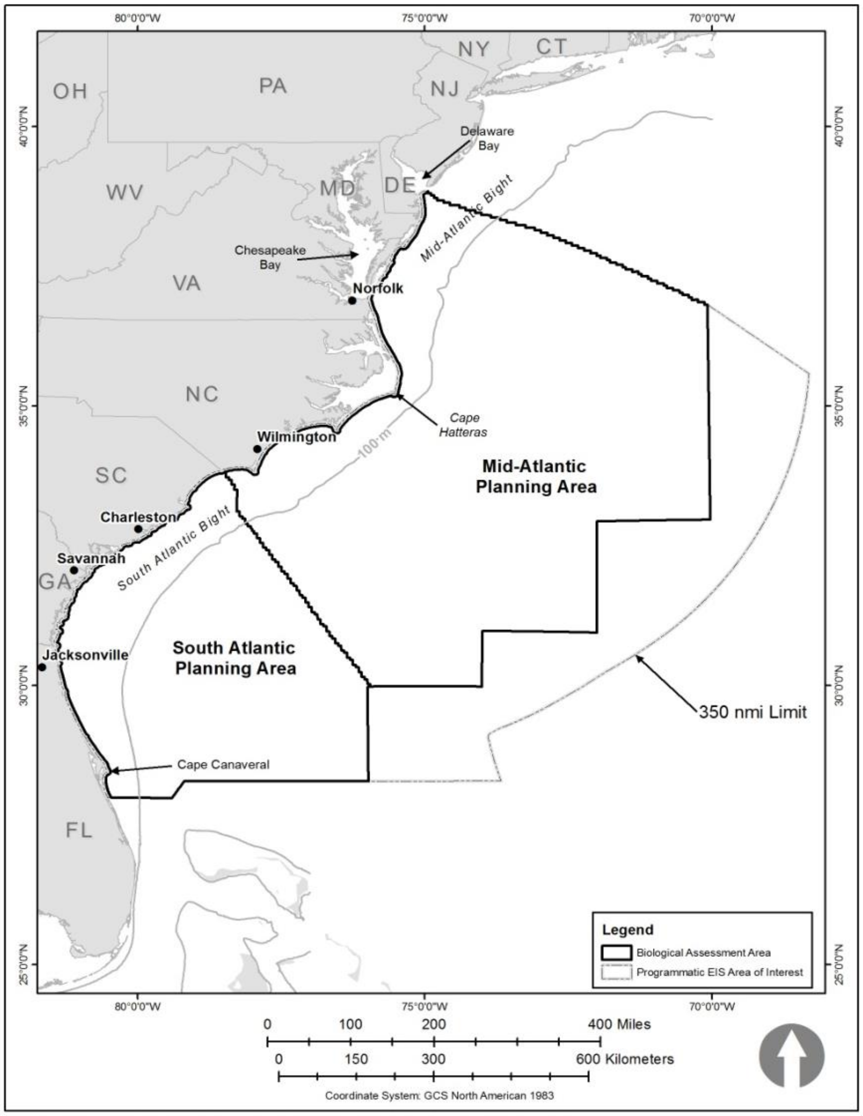
As sound travels through the ocean, the intensity associated with the wave front diminishes, or attenuates. In shallow waters of coastal regions and on continental shelves, sound speed profiles become influenced by surface heating and cooling, salinity changes, and water currents. As a result, these profiles tend to be irregular and unpredictable, and contain numerous gradients that last over short time and space scales. This decrease in intensity is referred to as propagation loss, also commonly called transmission loss. In general, in a homogeneous lossless medium, sound intensity decreases as the square of the range due to simple spherical spreading. In other words, a source level of 235 dB will have decreased in intensity to a received level of 175 dB after about 914 meters (1,000 yards).

Action Area

The Action Area for this biological opinion encompasses the marine and coastal waters off the states of New Jersey to Delaware and within and adjacent to BOEM's Mid-and South Atlantic Planning Areas (See Figure 4). The Mid- and South Atlantic Planning Areas includes federal waters from the mouth of the Delaware River (38^{o}51'N) to south of Cape Canaveral, Florida (28^o N), from the coastline (excluding estuaries) to 350 nm (638 km) from shore See Figure 2). Within the Action Area, activities conducted under the 3 program areas (oil and gas, renewable energy and marine minerals) can occur within the following areas:

- Oil and gas exploration and development -- these surveys and sampling activities are expected to occur anywhere within federal waters from 3 to 350 nm (6458 km) from shore;
- Renewable energy development --these surveys are expected to occur within 25 nmi (46.3 km) from shore or within 100 m (328 ft) of water;
- Marine mineral activities (e.g., sand and gravel mining) -- these activities are expected to occur within 10-30 m depths (33 to 98 ft) mostly within existing borrow areas along the eastern seaboard but could occur within previously undiscovered borrow areas; and
- Atlantic Wind Connection Transmission Cable -- these HRG surveys are expected to occur within waters within 25 nmi (46.3 km) from shore or within 100 m (328 ft) of water from New Jersey to Virginia.

Figure 4: Action Area Encompassing the Mid- and South Atlantic Planning Areas and adjacent state and Federal Waters. For Atlantic Wind Transmission Cable see Figure 3.



Status of Listed Resources

NMFS has determined that the following species and critical habitat designations may occur in this action area for this consultation:

Blue whale	*Balaenoptera musculus*	Endangered
Fin whale	*Balaenoptera physalus*	Endangered
Humpback whale	*Megaptera novaeangliae*	Endangered
North Atlantic right whale	*Eubalaena glacialis*	Endangered
Sei whale	*Balaenoptera borealis*	Endangered
Sperm whale	*Physeter macrocephalus*	Endangered
Green sea turtle	*Chelonia mydas*	Threatened
Hawksbill sea turtle	*Eretmochelys imbricata*	Endangered
Kemp’s ridley sea turtle	*Lepidochelys kempii*	Endangered
Leatherback sea turtle	*Dermochelys coriacea*	Endangered
Loggerhead sea turtle (Northwest Atlantic)	*Caretta caretta*	Threatened
Smalltooth sawfish	*Pristis pectinata*	Endangered
Shortnose sturgeon	*Acipenser brevirostrum*	Endangered
Atlantic sturgeon	*Acipenser oxyrinchus oxyrinchus*	Endangered/Threatened

Critical habitat has also been designated for the North Atlantic right whale in the Atlantic Ocean in Cape Cod Bay, Great South Channel, and off Georgia and Florida (50 CFR 226.203). Critical habitat for green sea turtles has been designated on Culebra Island, Puerto Rico (50 CFR 226.208), for hawksbill sea turtles on Mona and Monita Islands, Puerto Rico (50 CFR 226.209), and for leatherback sea turtles on Sandy Point on Saint Croix in the U.S. Virgin Islands (50 CFR 226.207).

Species Not Considered Further in this Opinion

As described in the *Approach to the Assessment section* of this Opinion, NMFS uses two criteria to identify those endangered or threatened species or critical habitat that are not likely to be adversely affected by the proposed G&G activities in the Action Area from 2013 to 2020. The first criterion was *exposure* or some reasonable expectation of a co-occurrence between one or more potential stressors associated with G&G activities and a particular listed species or designated critical habitat: if we conclude that a listed species or designated critical habitat is not likely to be exposed to G&G activities, we must also conclude that the species or critical habitat is not likely to be adversely affected by those activities. The second criterion is the probability of a *response* given exposure, which considers *susceptibility*: species or elements of critical habitat that may be exposed to sound transmissions from seismic or electromechanical equipment, for example, but are likely to be unaffected by the transmissions (at sound pressure levels or frequencies they are likely to be exposed to) are also not likely to be adversely affected by use of the equipment. We applied these criteria to the species listed at the beginning of this section; this subsection summarizes the results of those evaluations.

Smalltooth Sawfish

Smalltooth sawfish are tropical, marine and estuarine fish that inhabit shallow waters of inshore bars, mangrove edges, and seagrass beds, although they are occasionally found in deeper coastal waters (NMFS 2000). Historically, this species was common in the shallow waters of the Gulf of Mexico and along the eastern seaboard of the United States to North Carolina (rare sightings of this sawfish occurred as far north as New York). Their current range is limited to peninsular Florida, where they are only found with any regularity off the extreme southern portion of the peninsula (off Everglades National Park and Florida Bay). Because of their current distribution, smalltooth sawfish might only be exposed to the G&G activities proposed off the coast of Florida; however, the Action Area remains north of smalltooth sawfish primary distribution. Therefore, smalltooth sawfish are not likely to be exposed to the proposed G&G activities, and, therefore, are not likely to be adversely affected by the proposed activities.

Shortnose Sturgeon

Shortnose sturgeon occur along the Atlantic Coast of North America, from the St. John River in Canada to the St. Johns River in Florida. Shortnose sturgeon are known to occur at depths of up to 30m (98.4 ft) but are generally found in waters less than 20m (65.5 ft) (Dadswell *et al.* 1984; Dadswell 1979), however, a minimum depth of 0.6m (approximately 2 feet) is necessary for the unimpeded swimming by adults.

Because they rarely leave their natal rivers, Kieffer and Kynard (1993) considered shortnose sturgeon to be freshwater amphidromous (*i.e.* adults spawn in freshwater but regularly enter saltwater habitats during their life). Telemetry data, however, do indicate that shortnose sturgeon do make localized coastal migrations although these migrations are not the significant marine migrations seen in Atlantic sturgeon.

Interbasin movements have been documented among rivers within the Gulf of Maine (GOM) and between the GOM and the Merrimack and between the Connecticut and Hudson rivers. These rivers are outside the Action Area for this consultation. Within the Action Area interbasin transfers have been document between the Delaware River and Chesapeake Bay and among rivers in the Southeast. Genetic analyses suggest that individual shortnose sturgeon migrate between some of these populations each generation (Quattro *et al.* 2002, Wirgin *et al.* 2005). At the northern end of the species' distribution, the highest rate of gene flow (which suggests migration) occurs between the Kennebec and Androscoggin Rivers which empty into the Gulf of Maine. At the southern end of the species' distribution, populations south of the Pee Dee River within North and South Carolina appear to exchange between 1 and 10 individuals per generation (30 years), with the highest rates of exchange within the Ogeechee River in Georgia and the Delaware River (Wirgin *et al.* 2005). Wirgin *et al.* (2005) concluded that rivers separated by more than 400 km were connected by very little migration while rivers separated by no more than 20 km (such as the rivers flowing into coastal South Carolina) would experience high migration rates. At the geographic center of the shortnose sturgeon range, there is a 400 km stretch of river with no known populations occurring from the Delaware River, New Jersey to Cape Fear River, North Carolina (Kynard 1997). The shortnose sturgeon that are known to occur in the Chesapeake Bay may be transients from the Delaware River via the Chesapeake and Delaware Canal (Skjeveland *et al.* 2000, Welsh *et al.* 2002) or remnants of a population in the Potomac River and not the result of regular coastal migrations.

Based on the distribution of shortnose sturgeon populations, the low expected rate of migrations outside of natal rivers and preference for shallow water, the probability of shortnose sturgeon exposure to G&G activities within the Action Area is expected to be so low as to be discountable. As such shortnose sturgeon are not likely to be adversely affected by the proposed activities and we will not consider them further in this Opinion.

Critical Habitat

Critical habitat has also been designated for the North Atlantic right whale in Cape Cod Bay and the Great South Channel off Massachusetts, and off Georgia and Florida (50 CFR 226.203). Critical habitat for green sea turtles has been designated on Culebra Island, Puerto Rico (50 CFR226.208), for hawksbill sea turtles on Mona and Monita Islands, Puerto Rico (50 CFR 226.209), and for leatherback sea turtles on Sandy Point on Saint Croix in the U.S. Virgin Islands (50 CFR 226.207).

Based on the best scientific and commercial data available, critical habitat that has been designated for North Atlantic right whales off the coast of Massachusetts, as well as critical habitat designated for green sea turtles, hawksbill sea turtles, and leatherback sea turtles is outside of the area that might be exposed to the proposed G&G activities. As a result, we conclude that the proposed action will not affect these areas of designated critical habitat. Therefore, this critical habitat will not be considered further in this opinion.

Critical habitat has been designated for North Atlantic right whales in the southeast U.S. (off Georgia and Florida) encompassing an area extending from 31^{o}15 N to 30^{o}15 N from the coast out to 28 km (15 nmi), and the area from 28^{o}00 N to 30^{o}15 N from the coast out to 9 km (5 nm) to protect calving and nursing right whales and their young.

Under the proposed action considered in this consultation, no surveys using airguns would be authorized from November 15 to April 15, annually, within southeast right whale critical habitat when right whales are present, to limit the effects of the proposed airgun survey activities on North Atlantic right whales. For non-airgun HRG surveys within southeast North Atlantic right whale critical habitat, BOEM proposes to authorize surveys using electromechanical equipment during right whale calving and nursing season. These surveys would generally operate at frequencies above the audibility range of North Atlantic right whales (30 kHz). Only surveys BOEM determines to be critical to program activities can be authorized to use frequencies below 30 kHz. This determination will consider whether survey planning could have scheduled survey activities outside of the calving and nursing season and how the particular survey fills a critical need of the program. For these surveys, BOEM will only authorize the use of sound source during daylight hours (Marine Minerals Program) or use of sound sources above 200 kHz at night and sound sources within the audibility range of North Atlantic right whales during daylight hours (Renewable Energy Program). Only 1-2 surveys that may require the use of sound sources below 30 kHz are anticipated to occur per year within North Atlantic right whale critical habitat in the southeast during calving and nursing season.

Neither seismic airgun nor non-airgun HRG survey activities affect the essential features (temperatures, bathymetry and calm water) of this critical habitat; therefore North Atlantic right whale critical habitat will not be affected by the proposed G&G activities and we will not consider southeastern right whale critical habitat further in this Opinion. We will, however, consider further in this opinion the exposure, responses and the risks of those responses given exposure to individual North Atlantic right whales that occur within critical habitat during survey activities.

Introduction to Status of Listed Species

The rest of this section of our Opinion consists of narratives for each of the threatened and endangered species that occur in the Action Area and that may be adversely affected by the proposed G&G activities along the Atlantic Coast of the United States from 2013 to 2020. In each narrative, we present a summary of information on the distribution and population structure of each species to provide a foundation for the exposure analyses that appear later in this Opinion. Then we summarize information on the threats to the species and the species' status given those threats to provide points of reference for the jeopardy determinations we make later in this Opinion. That is, we rely on a species' status and trend to determine whether or not an action's direct or indirect effects are likely to increase the species' probability of becoming extinct.

After the Status subsection of each narrative, we present information on the diving and social behavior of the different species because that behavior helps determine whether monitoring efforts during G&G activities are likely to detect each species. We also summarize information on the vocalizations and hearing of the different species because that background information lays the foundation for our assessment of how the different species are likely to respond to sounds produced by G&G activities.

More detailed background information on the status of these species and critical habitat can be found in a number of published documents including status reviews and recovery plans for blue whales (NMFS 1998a), fin whales (NMFS 2007, 2010a), fin and sei whale (NMFS 1998b, NMFS 2007, NMFS 2012), humpback whales (NMFS 1991a), sperm whales (NMFS 2010b) and a status report on large whales prepared by Perry *et al.* (1999). Sea turtle background information can be found in recovery plans (NMFS and USFWS 1998a, 1998b, 1998c, 1998d, and 1998e) 5-year reviews (NMFS 2007) and status reviews (Conant et al 2009). More detailed information on Atlantic sturgeon can be found in the status review (Atlantic Sturgeon Status Review Team. 2007).

For the following species, the term "population" refers to groups of individuals whose patterns of increase or decrease in abundance over time are determined by internal dynamics (births resulting from sexual interactions between individuals in the group and deaths of those individuals) rather than external dynamics (immigration or emigration). This definition is a reformulation of definitions articulated by Cole (1957), Futuyma (1986) and Wells and Richmond (1995) and is more restrictive than those uses of 'population' that refer to groups of individuals that co-occur in space and time but do not have internal dynamics that determine whether the size of the group increases or decreases over time (see review by Wells and Richmond 1995). The definition we apply is important to section 7 consultations because such concepts as "population decline", "population collapse", "population extinction", and "population recovery" apply to the restrictive definition of "population" but do not explicitly apply to alternative definitions. As a result, we do not treat the different whale "stocks" recognized by the International Whaling Commission or other authorities as populations unless those distinctions were clearly based on demographic criteria. We do, however, acknowledge those "stock" distinctions in these narratives.

Blue Whale

Distribution

Blue whales are found along the coastal shelves of North America and South America (Rice 1974; Donovan 1984; Clarke 1980) in the North Pacific Ocean. In the North Pacific Ocean, blue whales occur in summer foraging areas in the Chukchi Sea, the Sea of Okhotsk, around the Aleutian Islands, and the Gulf of Alaska; in the eastern Pacific, they occur south to California; in the western Pacific, they occur south to Japan. Blue whales in the eastern Pacific winter from California south; in the western Pacific, they winter from the Sea of Japan, the East China and Yellow Seas, and the Philippine Sea (Gambell 1985).

In the western North Atlantic Ocean, blue whales are found from the Arctic to at least the mid-latitude waters of the North Atlantic (CeTAP 1982, Wenzel *et al.*1988, Yochem and Leatherwood 1985, Gagnon and Clark 1993). Blue whales have been observed frequently off eastern Canada, particularly in waters off Newfoundland, during the winter. In the summer month, they have been observed in Davis Strait (Mansfield 1985), the Gulf of St. Lawrence (from the north shore of the St. Lawrence River estuary to the Strait of Belle Isle), and off eastern Nova Scotia (Sears *et al.* 1987). In the eastern North Atlantic Ocean, blue whales have been observed off the Azores Islands, although Reiner *et al.* (1993) do not consider them common in that area.

In 1992, the U.S. Navy conducted an extensive acoustic survey of the North Atlantic using the Integrated Underwater Surveillance System’s fixed acoustic array system (Clark 1995). Concentrations of blue whale sounds were detected in the Grand Banks off Newfoundland and west of the British Isles. In the lower latitudes, one blue whale was tracked acoustically for 43 days, during which time the animal traveled 1400 nautical miles around the western North Atlantic from waters northeast of Bermuda to the southwest and west of Bermuda (Gagnon and Clark 1993).

In the North Pacific Ocean, blue whales have been recorded off the island of Oahu in the main Hawaiian Islands and off Midway Island in the western edge of the Hawaiian Archipelago (Barlow *et al.* 1994b; Northrop *et al.* 1971; Thompson and Friedl 1982), although blue whales are rarely sighted in Hawaiian waters and have not been reported to strand in the Hawaiian Islands. Nishiwaki (1966) reported that blue whales occur in the Aleutian Islands and in the Gulf of Alaska, although blue whales have not been observed off Alaska since 1987 (Leatherwood *et al.* 1982; Stewart *et al.* 1987; Forney and Brownell 1996). No distributional information exists for the western region of the North Pacific.

In the eastern tropical Pacific Ocean, the Costa Rica Dome appears to be important for blue whales based on the high density of prey (euphausiids) available in the Dome and the number of blue whales that appear to reside there (Reilly and Thayer 1990). Blue whales have been sighted in the Dome area in every season of the year, although their numbers appear to be highest from June through November.

Blue whales have also been reported year-round in the northern Indian Ocean, with sightings in the Gulf of Aden, Persian Gulf, Arabian Sea, and across the Bay of Bengal to Burma and the Strait of Malacca (Mizroch *et al.* 1984). The migratory movements of these whales are unknown.

Historical catch records suggest that "true" blue whales and "pygmy" blue whale (*B. m. brevicauda*) may be geographically distinct (Brownell and Donaghue 1994, Kato *et al.* 1995). The distribution of the "pygmy" blue whale is north of the Antarctic Convergence, while that of the "true" blue whale is south of the Convergence in the austral summer (Kato *et al.* 1995). "True" blue whales occur mainly in the higher latitudes, where their distribution in mid-summer overlaps with that of the minke whale (*Balaenoptera acutorostrata*). During austral summers, "true" blue whales are found close to edge of Antarctic ice (south of 58° S) with concentrations between 60°-80° E and 66°-70° S (Kasamatsu *et al.* 1996).

Population Structure

At least three subspecies of blue whales have been identified based on body size and geographic distribution (*B. musculus intermedia*, which occurs in the higher latitudes of the Southern Oceans, *B. m. musculus*, which occurs in the Northern Hemisphere, and *B. m. brevicauda* which occurs in the mid-latitude waters of the southern Indian Ocean and north of the Antarctic convergence), but this consultation will treat them as a single entity. Readers who are interested in these subspecies will find more information in Gilpatrick *et al.* (1997), Kato *et al.* (1995), Omura *et al.* (1970) and Ichihara (1966).

In addition to these subspecies, the International Whaling Commission's Scientific Committee has formally recognized one blue whale population in the North Pacific (Donovan 1991), although there is increasing evidence that there may be more than one blue whale population in the Pacific Ocean (Gilpatrick *et al.* 1997, Barlow *et al.* 1995, Mizroch *et al.* 1984a, Ohsumi and Wada 1974). For example, studies of the blue whales that winter off Baja California and in the Gulf of California suggest that these whales are morphologically distinct from blue whales of the western and central North Pacific (Gilpatrick *et al.* 1997), although these differences might result from differences in the productivity of their foraging areas more than genetic differences (the southern whales forage off California; Sears *et al.*1987; Barlow *et al.*1997; Calambokidis *et al.* 1990).

A population or “stock” of endangered blue whales occurs in waters surrounding the Hawaiian archipelago (from the main Hawaiian Islands west to at least Midway Island), although blue whales are rarely reported from Hawaiian waters. The only reliable report of this species in the central North Pacific was a sighting made from a scientific research vessel about 400 km northeast of Hawai’i in January 1964 (NMFS 1998). However, acoustic monitoring has recorded blue whales off Oahu and the Midway Islands much more recently (Barlow *et al.* 1994, McDonald and Fox 1999, Northrop *et al.* 1971; Thompson and Friedl 1982).

The recordings made off Oahu showed bimodal peaks throughout the year, suggesting that the animals were migrating into the area during summer and winter (Thompson and Friedl 1982; McDonald and Fox 1999). Twelve aerial surveys were flown within 25 nm^2 of the main Hawaiian Islands from 1993-1998 and no blue whales were sighted. Nevertheless, blue whale vocalizations that have been recorded in these waters suggest that the occurrence of blue whales in these waters may be higher than blue whale sightings. There are no reports of blue whales stranding in Hawaiian waters.

The International Whaling Commission also groups all of the blue whales in the North Atlantic Ocean into one “stock” and groups blue whales in the Southern Hemisphere into six “stocks” (Donovan 1991), which are presumed to follow the feeding distribution of the whales.

Threats to the Species

Natural threats. Natural causes of mortality in blue whales are largely unknown, but probably include predation and disease (not necessarily in their order of importance). Blue whales are known to become infected with the nematode *Carricauda boopis* (Baylis 1920), which are also believed to have caused fin whales to die as a result of renal failure (Lambertsen 1986; see additional discussion under *Fin whales*). Killer whales and sharks are also known to attack, injure, and kill very young or sick fin and humpback whale and probably hunt blue whales as well (Perry *et al.* 1999).

Anthropogenic threats. Two human activities are known to threaten blue whales: whaling and shipping. Historically, whaling represented the greatest threat to every population of blue whales and was ultimately responsible for listing blue whales as an endangered species. As early as the mid-seventeenth century, the Japanese were capturing blue, fin, and other large whales using a fairly primitive open-water netting technique (Tønnessen and Johnsen 1982, Cherfas 1989). In 1864, explosive harpoons and steam-powered catcher boats were introduced in Norway, allowing the large-scale exploitation of previously unobtainable whale species. Before fin whales became the focus of whaling operations, populations of blue whales had already become commercially extinct (IWC 1995).

From 1910 to 1965, whalers killed about 9,500 blue whales in the North Pacific Ocean (NMFS 1998). Although the International Whaling Commission banned commercial whaling in the North Pacific in 1966, Soviet whaling fleets continued to hunt blue whales in the North Pacific for several years after the ban. Surveys conducted in these former-whaling areas in the 1980s and 1990s failed to find any blue whales (Forney and Brownell 1996). By 1967, Soviet scientists wrote that blue whales in the North Pacific Ocean (including the eastern Bering Sea and Prince William Sound) had been so overharvested by Soviet whaling fleets that some scientists concluded that any additional harvests were certain to cause the species to become extinct in the North Pacific (Latishev 2007). As its legacy, whaling has reduced blue whales to a fraction of their historic population size and, as a result, makes it easier for other human activities to push blue whales closer to extinction. Otherwise, whaling currently does not threaten blue whale populations.

In the Eastern North Pacific, ship strikes were implicated in the deaths of five blue whales, from 2004-2008 (Caretta *et al*. 2011). Four of these deaths occurred in 2007, the highest number recorded for any year. During 2004-2008, there were an additional eight injuries of unidentified large whales attributed to ship strikes. Several blue whales have been photographed in California with large gashes in their dorsal surface that appear to be from ship strikes (J. Calambokidis, pers. comm.). Blue whale mortality and injuries attributed to ship strikes in California waters averaged 1.0 per year for 2004-2008. 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 *et al.* 1983). Within the St. Lawrence Estuary, blue whales are believed to be affected by large amounts of recreational and commercial vessel traffic. Blue whales in the St. Lawrence appeared more likely to react to these vessels when boats made fast, erratic approaches or sudden changes in direction or speed (Edds and Macfarlane 1987, Macfarlane 1981). No confirmed ship strikes of blue whales have been recorded in the North Atlantic and Gulf of Mexico between 2006 and 2010 (Henry et al. 2012).

Status

Blue whales were listed as endangered under the ESA in 1973. Blue whales are listed as endangered on the IUCN Red List of Threatened Animals (Baillie and Groombridge 1996). They are also protected by the Convention on International Trade in Endangered Species of wild flora and fauna and the MMPA. Critical habitat has not been designated for blue whales.

It is difficult to assess the current status of blue whales because (1) there is no general agreement on the size of the blue whale population prior to whaling and (2) estimates of the current size of the different blue whale populations vary widely. We may never know the size of the blue whale population prior to whaling, although some authors have concluded that their population numbers about 200,000 animals before whaling. Similarly, estimates of the global abundance of blue whales are uncertain. Since the cessation of whaling, the global population of blue whales has been estimated to range from 11,200 to 13,000 animals (Maser *et al.* 1981; U. S. Department of Commerce 1983). These estimates, however, are more than 20 years old.

A lot of uncertainty surrounds estimates of blue whale abundance in the North Pacific Ocean. Barlow (1994) estimated the North Pacific population of blue whales to number between 1,400 and 1,900. Barlow and Calambokidis (1995) estimated the abundance of blue whales off California at 2,200 individuals. Wade and Gerrodette (1993) and Barlow *et al.* (1997) estimated there were a minimum of 3,300 blue whales in the North Pacific Ocean in the 1990s. Most recently, Calambokidis et al. (2010) estimated that the eastern North Pacific stock size is increasing at approximately 3% per year.

The size of the blue whale population in the North Atlantic is also uncertain. The population has been estimated to number from a few hundred individuals (Allen 1970; Mitchell 1974) to 1,000 to 2,000 individuals (Sigurjónsson 1995). Gambell (1976) estimated that there were between 1,100 and 1,500 blue whales in the North Atlantic before whaling began and Braham (1991) estimated there were between 100 and 555 blue whales in the North Atlantic during the late 1980s and early 1990s. Sears *et al.* (1987) identified over 300 individual blue whales in the Gulf of St. Lawrence, which provides a minimum estimate for their population in the North Atlantic, and some speculate that there may be between 400 and 600 blue whales in the western North Atlantic (Mitchell 1974, Waring et al. 2011). Sigurjónsson and Gunnlaugson (1990) concluded that the blue whale population had been increasing since the late 1950s and argued that the blue whale population off western and southwestern Iceland had increased at an annual rate of about 5 percent between 1969 and 1988, although the level of confidence we can place in these estimates is low.

Estimates of the number of blue whales in the Southern Hemisphere range from 5,000 to 6,000 (review by Yochem and Leatherwood 1985) with an average rate of increase that has been estimated at between 4 and 5 percent per year. Butterworth *et al.* (1993), however, estimated the Antarctic population at 710 individuals. More recently, Stern (2001) estimated the blue whale population in the Southern Ocean at between 400 and 1,400 animals (C.V. 0.4). The pygmy blue whale population has been estimated at 6,000 individuals (Yochem and Leatherwood 1985).

The information available on the status and trend of blue whales do not allow us to reach any conclusions about the extinction risks facing blue whales as a species, or particular populations of blue whales. With the limited data available on blue whales, we do not know whether these whales exist at population sizes large enough to avoid demographic phenomena that are known to increase the extinction probability of species that exist as "small" populations (that is, "small" populations experience phenomena such as demographic stochasticity, inbreeding depression and Allee effects, among others, that cause their population size to become a threat in and of itself) or if blue whales are threatened more by exogenous threats such as anthropogenic activities (primarily whaling, entanglement, and ship strikes) or natural phenomena (such as disease, predation, or changes in the distribution and abundance of their prey in response to changing climate).

Diving and Social Behavior

Generally, blue whales make 5-20 shallow dives at 12-20 second intervals followed by a deep dive of 3-30 minutes (Mackintosh 1965; Leatherwood *et al.* 1976; Maser *et al.* 1981; Yochem and Leatherwood 1985; Strong 1990; Croll *et al.* 1999). Croll *et al.* (1999) found that the dive depths of blue whales foraging off the coast of California during the day averaged 132 m (433 ft) with a maximum recorded depth of 204 m (672 ft) and a mean dive duration of 7.2 minutes. Nighttime dives are generally less than 50 m (165 ft) in depth (Croll *et al.* 1999).

Blue whales are usually found swimming alone or in groups of two or three (Ruud 1956, Slijper 1962, Nemoto 1964, Mackintosh 1965, Pike and MacAskie 1969, Aguayo 1974). However, larger foraging aggregations and aggregations mixed with other species like fin whales are regularly reported (Schoenherr 1991, Fiedler *et al.* 1998). Little is known of the mating behavior of blue whales.

Vocalizations and Hearing

The vocalizations that have been identified for blue whales include a variety of sounds described as low frequency moans or long pulses (Cummings and Thompson 1971, 1977; Edds 1982, Thompson and Friedl 1982; Edds-Walton 1997). Blue whales produce a variety of low frequency sounds in the 10-100 Hz band (Cummings and Thompson 1971, Edds 1982, Thompson and Friedl 1982, McDonald *et al.* 1995, Clark and Fristrup 1997, Rivers 1997). The most typical signals are very long, patterned sequences of tonal infrasonic sounds in the 15-40 Hz range. The sounds last several tens of seconds. Estimated source levels are as high as 180-190 dB (Cummings and Thompson 1971). Ketten (1997) reports the frequencies of maximum energy between 12 and 18 Hz. In temperate waters, intense bouts of long patterned sounds are very common from fall through spring, but these also occur to a lesser extent during the summer in high latitude feeding areas. Short sequences of rapid calls in the 30-90 Hz band are associated with animals in social groups. The seasonality and structure of long patterned sounds suggest that these sounds are male displays for attracting females, competing with other males, or both. The context for the 30-90 Hz calls suggests that they are communicative but not related to a reproductive function. Vocalizations attributed to blue whales have been recorded in presumed foraging areas, along migration routes, and during the presumed breeding season (Beamish and Mitchell 1971; Cummings and Thompson 1971, 1977, 1994; Cummings and Fish 1972; Thompson *et al.* 1996; Rivers 1997; Tyack and Clark 1997; Clark *et al.* 1998).

Blue whale moans within the low frequency range of 12.5-200 Hz, with pulse duration up to 36 seconds, have been recorded off Chile (Cummings and Thompson 1971). A short, 390 Hz pulse also is produced during the moan. One estimate of the overall source level was as high as 188 dB, with most energy in the 1/3-octave bands centered at 20, 25, and 31.5 Hz, and also included secondary components estimates near 50 and 63 Hz (Cummings and Thompson 1971).

Direct studies of large whale hearing have not been conducted for most species of large whales, but it is assumed that they can hear the same frequencies that they produce (low) and are likely most sensitive to that frequency range (D. R. Ketten, 1997; Richardson, Jr., Malme, & Thomson, 1995). A more recent study involving blue whales off California observed responses to mid-frequency sonar signals in the 1-8 kHz range (Melcón et al 2012). This study challenges our assumptions about the hearing ranges of large whales.

As with other vocalizations produced by baleen whales, the function of blue whale vocalizations is unknown, although there are numerous hypotheses (which include: maintenance of inter-individual distance, species and individual recognition, contextual information transmission, maintenance of social organization, location of topographic features, and location of prey resources; see the review by Thompson *et al.* 1992 for more information on these hypotheses). Responses to conspecific sounds have been demonstrated in a number of mysticetes, and there is no reason to believe that blue whales do not communicate similarly (Edds-Walton 1997). The low-frequency sounds produced by blue whales can, in theory, travel long distances, and it is possible that such long-distance communication occurs (Payne and Webb 1971, Edds-Walton 1997). The long-range sounds may also be used for echolocation in orientation or navigation (Tyack 1999).

Cetaceans have an auditory anatomy that follows the basic mammalian pattern, with some modifications to adapt to the demands of hearing in the sea. The typical mammalian ear is divided into the outer ear, middle ear, and inner ear. The outer ear is separated from the inner ear by the tympanic membrane, or eardrum. In terrestrial mammals, the outer ear, eardrum, and middle ear function to transmit airborne sound to the inner ear, where the sound is detected in a fluid. Since cetaceans already live in a fluid medium, they do not require this matching, and thus do not have an air-filled external ear canal. The inner ear is where sound energy is converted into neural signals that are transmitted to the central nervous system via the auditory nerve. Acoustic energy causes the basilar membrane in the cochlea to vibrate. Sensory cells at different positions along the basilar membrane are excited by different frequencies of sound (Tyack 1999). Baleen whales have inner ears that appear to be specialized for low-frequency hearing. In a study of the morphology of the mysticete auditory apparatus, Ketten (1997) hypothesized that large mysticetes have acute infrasonic hearing.

Fin Whale

Distribution

Fin whales are distributed widely in every ocean except the Arctic Ocean. In the North Pacific Ocean, fin whales occur in summer foraging areas in the Chukchi Sea, the Sea of Okhotsk, around the Aleutian Islands, and the Gulf of Alaska; in the eastern Pacific, they occur south to California; in the western Pacific, they occur south to Japan. Fin whales in the eastern Pacific winter from California south; in the western Pacific, they winter from the Sea of Japan, the East China and Yellow Seas, and the Philippine Sea (Gambell 1985).

In the North Atlantic Ocean, fin whales occur in summer foraging areas from the coast of North America to the Arctic, around Greenland, Iceland, northern Norway, Jan Meyers, Spitzbergen, and the Barents Sea. In the western Atlantic, they winter from the edge of sea ice south to the Gulf of Mexico and the West Indies. In the eastern Atlantic, they winter from southern Norway, the Bay of Biscay, and Spain with some whales migrating into the Mediterranean Sea (Gambell 1985).

In the Southern Hemisphere, fin whales are distributed broadly south of 50° S in the summer and migrate into the Atlantic, Indian, and Pacific Oceans in the winter, along the coast of South America (as far north as Peru and Brazil), Africa, and the islands in Oceania north of Australia and New Zealand (Gambell 1985).

Fin whales are common off the Atlantic coast of the United States in waters immediately off the coast seaward to the continental shelf (about the 1,000-fathom contour). In this region, they tend to occur north of Cape Hatteras, where they accounted for about 46 percent of the large whales observed in surveys conducted between 1978 and 1982. During the summer months, fin whales in this region tend to congregate in feeding areas between 41°20'N and 51°00'N, from shore seaward to the 1,000-fathom contour.

In the Atlantic Ocean, Clark (1995) reported a general southward pattern of fin whale migration in the fall from the Labrador and Newfoundland region, south past Bermuda, and into the West Indies. The overall distribution may be based on prey availability, and fin whales are found throughout the action area for this consultation in most months of the year. This species preys opportunistically on both invertebrates and fish (Watkins *et al.* 1984). They feed by filtering large volumes of water for the associated prey. Fin whales are larger and faster than humpback and right whales and are less concentrated in nearshore environments.

Population Structure

Fin whales have two recognized subspecies: *Balaenoptera physalus physalus* (Linnaeus 1758) occurs in the North Atlantic Ocean while *B. p***.** *quoyi* (Fischer 1829) occurs in the Southern Ocean. Globally, fin whales are sub-divided into three major groups: Atlantic, Pacific, and Antarctic. Within these major areas, different organizations use different population structure.

In the North Atlantic Ocean, the International Whaling Commission recognizes seven management units or “stocks” of fin whales: (1) Nova Scotia, (2) Newfoundland-Labrador, (3) West Greenland, (4) East Greenland-Iceland, (5) North Norway, (6) West Norway-Faroe Islands, and (7) British Isles-Spain-Portugal. In addition, the population of fin whales that resides in the Ligurian Sea, in the northwestern Mediterranean Sea, is believed to be genetically distinct from other fin whales populations (as used in this Opinion, “populations” are isolated demographically, meaning, they are driven more by internal dynamics — birth and death processes — than by the geographic redistribution of individuals through immigration or emigration. Some usages of the term “stock” are synonymous with this definition of “population” while other usages of “stock” are not).

In the North Pacific Ocean, the International Whaling Commission recognizes two “stocks”: (1) East China Sea and (2) rest of the North Pacific (Donovan, 1991). However, Mizroch *et al.* (1984) concluded that there were five possible “stocks” of fin whales within the North Pacific based on histological analyses and tagging experiments: (1) East and West Pacific that intermingle around the Aleutian Islands; (2) East China Sea; (3) British Columbia; (4) Southern-Central California to Gulf of Alaska; and (5) Gulf of California. Based on genetic analyses,

Berube *et al.* (1998) concluded that fin whales in the Sea of Cortez represent an isolated population that has very little genetic exchange with other populations in the North Pacific Ocean (although the geographic distribution of this population and other populations can overlap seasonally). They also concluded that fin whales in the Gulf of St. Lawrence and Gulf of Maine are distinct from fin whales found off Spain and in the Mediterranean Sea. Regardless of how different authors structure the fin whale population, mark-recapture studies have demonstrated that individual fin whales migrate between management units (Mitchell 1974; Gunnlaugsson and Sigurjónsson 1989), which suggests that these management units are not geographically isolated populations.

In its final recovery plan for fin whales, NMFS recognized three populations in U.S. Pacific waters: Alaska (Northeast Pacific), California/Oregon/Washington, and Hawaii (NMFS 2010).

Threats to the Species

Natural threats. Natural sources and rates of mortality are largely unknown, but Aguilar and Lockyer (1987) suggest annual natural mortality rates may range from 0.04 to 0.06. Although these results are based on studies of fin whales in the northeast Atlantic, there are no comparable estimates for fin whales in the Pacific Ocean. The occurrence of the nematode *Crassicauda boopis* appears to increase the potential for kidney failure in fin whales and may be preventing some fin whale stocks from recovering from whaling (Lambertsen 1992, as cited in Perry *et al.* 1999). Killer whale or shark attacks may injure or kill very young or sick whales (Perry *et al.* 1999, Tomilin 1967).

Anthropogenic threats. Three human activities are known to threaten fin whales: whaling, commercial fishing, and shipping. Historically, whaling represented the greatest threat to every population of fin whales and was ultimately responsible for listing fin whales as an endangered species. As early as the mid-seventeenth century, the Japanese were capturing fin, blue (*Balaenoptera musculus*), and other large whales using a fairly primitive open-water netting technique (Tønnessen and Johnsen 1982, Cherfas 1989). In 1864, explosive harpoons and steam-powered catcher boats were introduced in Norway, allowing the large-scale exploitation of previously unobtainable whale species. After blue whales were depleted in most areas, fin whales became the focus of whaling operations and more than 700,000 fin whales were landed in the Southern Hemisphere alone between 1904 and 1979 (IWC 1995).

As its legacy, whaling has reduced fin whales to a fraction of their historic population size and, as a result, makes it easier for other human activities to push fin whales closer to extinction. Otherwise, whaling currently does not threaten every fin whale population, although it may threaten specific populations.

From 1904 to 1975, the International Whaling Commission estimates that 703,693 fin whales were captured and killed in Antarctic whaling operations (IWC 1990). Whaling in the Southern Oceans originally targeted humpback whales, but by 1913, those whales had become rare so whalers shifted their focus to fin and blue whales (Mizroch *et al.* 1984b). From 1911 to 1924, whalers killed 2,000–5,000 fin whales each year. After the introduction of factory whaling ships in 1925, the number of whales killed each year increased substantially: from 1931 to 1972, whalers killed about 511,574 fin whales (Kawamura 1994). In 1937 alone, whalers are reported to have killed more than 28,000 fin whales. From 1953 to 1961, the number of fin whales killed each year averaged around 25,000. In 1962, whalers appeared to shift their focus to sei whale as fin whales became scarce. By 1974, whalers killed fewer than 1,000 fin whales.

Recently released Soviet whaling records indicate a discrepancy between reported and actual fin whale catch numbers by whalers from the former USSR in southern waters between 1947 and 1980 (Zemsky *et al.* 1995). The former USSR previously reported 52,931 whales caught; however, the data that was released recently suggests that only 41,984 were killed.

In the Antarctic Ocean, fin whales are hunted by Japanese whalers for its scientific whaling program under an Antarctic Special Permit. Japan started killing fin whales in its 2005–2006 program season and increased its target from 10 to 50 fin whales for the next twelve seasons beginning with the 2007/2008 season. Japan took zero fin whales in the 2007/2008 season and one in the 2008/2009 season (NMFS 2010a); however, in 2009 and the 2009/2010 seasons Japan killed a total of one fin whale in Antarctica (IWC 2010).

In the Atlantic Ocean fin whales are also hunted in subsistence fisheries off West Greenland and Iceland. In 2004, 5 males and 6 females were killed and landed; 2 other fin whales were struck and lost in the same year off West Greenland. In 2003 2 males and 4 females were landed and 2 other fin whales were struck and lost off West Greenland (IWC 2005). Between 2003 and 2007, the IWC set a catch limit of up to 19 fin whales in the West Greenland subsistence fishery (IWC 2005), however, the IWC's Scientific Committee recommended limiting the number of fin whale killed in this fishery to 1 to 4 individuals until accurate population estimates are produced. Seven fin whales were killed, 2 struck and lost and 1 reported killed as an infraction in 2009 and the 2009/2010 seasons combined (IWC 2010). Another 125 fin whales with 2 reported as infractions were killed off Iceland in 2009 and the 2009/2010 seasons combined (IWC 2010).

Despite anecdotal observations from fishermen which suggest that large whales swim through their nets rather than get caught in them (NMFS 2000), fin whales have been entangled by fishing gear off Newfoundland and Labrador in small numbers: a total of 14 fin whales are reported to have been captured in coastal fisheries in those two provinces between 1969 and 1990 (Lien 1994, Perkins and Beamish

1979). Of these 14 fin whales, 7 are known to have died as a result of that capture, although most of the animals that died were less than 15 meters in length (Lien 1994). Between 1999 and 2005, there were 10 confirmed reports of fin whales being entangled in fishing gear along the Atlantic Coast of the U.S. and the Maritime Provinces of Canada (Cole *et al.* 2005, Nelson *et al.* 2007). Of these reports, fin whales were injured in 1 of the entanglements and killed in 3 entanglements. These data suggest that, despite their size and strength, fin whales are likely to be entangled and, in some cases, killed by gear used in modern fisheries.

Fin whales are also killed and injured in collisions with vessels more frequently than any other whale. Of 92 fin whales that stranded along the Atlantic Coast of the U.S. between 1975 and 1996, 31 (33%) showed evidence of collisions with ships (Laist *et al.* 2001). Between 1999 and 2005, there were 15 reports of fin whales being struck by vessels along the Atlantic Coast of the U.S. and the Maritime Provinces of Canada (Cole *et al.* 2005, Nelson *et al.* 2007). Of these reports, 13 were confirmed as ship strikes which were reported as having resulted in the death of 11 fin whales.

Ship strikes were identified as a known or potential cause of death in 8 (20%) of 39 fin whales that stranded on the coast of Italy in the Mediterranean Sea between 1986 and 1997 (Laist *et al.* 2001). Throughout the Mediterranean Sea, 46 of the 287 fin whales that are recorded to have stranded between 1972 and 2001 were confirmed to have died from injuries sustained by ship strikes (Panigada *et al.* 2006). Most of these fin whales (n = 43), were killed between 1972 and 2001 and the highest percentage (37 of 45 or ~82%) were killed in the Ligurian Sea and adjacent waters, where the Pelagos Sanctuary for Marine Mammals was established. In addition to these ship strikes, there are numerous reports of fin whales being injured as a result of ship strikes off the Atlantic coast of France and the United Kingdom (Jensen and Silber 2003).

Status

Fin whales were listed as endangered under the ESA in 1970. In 1976, the IWC protected fin whales from commercial whaling (Allen 1980). Fin whales are listed as endangered on the IUCN Red List of Threatened Animals (Baillie and Groombridge 1996). They are also protected by the Convention on International Trade in Endangered Species of wild flora and fauna and the MMPA. Critical habitat has not been designated for fin whales.

It is difficult to assess the current status of fin whales because (1) there is no general agreement on the size of the fin whale population prior to whaling and (2) estimates of the current size of the different fin whale populations vary widely (NMFS 2007). We may never know the size of the fin whale population prior to whaling. Based on sighting and CPUE data and a population model, Ohsumi and Wada (1974) estimated that there were 8,000–11,000 fin whales in the eastern North Pacific in 1973. The most current (1979) estimate of the population size of fin whales in the Southern Ocean is 85,200 (no coefficient of variance or confidence interval was provided) based on the history of catches and trends in catches per unit of effort (IWC 1979).

According to the fin whale recovery plan (NMFS 2010), the best abundance estimate available for the Western North Atlantic stock is 2,269 (CV = 0.37) from August 2006 with a minimum population estimate of 1,678 (Waring *et al.* 2009). However, based on data produced by surveys conducted between 1978-1982 and other data gathered between 1966 and 1989, Hain *et al.* (1992) estimated that the population of fin whales in the western North Atlantic Ocean (specifically, between Cape Hatteras, North Carolina, and Nova Scotia) numbered about 1,500 whales in the winter and 5,000 whales in the spring and summer. Because authors do not always reconcile "new" estimates with earlier estimates, it is not clear whether the current "best" estimate represents a refinement of the estimate that was based on older data or whether the fin whale population in the North Atlantic has declined by about 50% since the early 1980s.

The East Greenland-Iceland fin whale population was estimated at 10,000 animals (95 % confidence interval = 7,600 - 14,200), based on surveys conducted in 1987 and 1989 (Buckland *et al.* 1992). The number of eastern Atlantic fin whales, which includes the British Isles-Spain-Portugal population, has been estimated at 17,000 animals (95% confidence interval = 10,400 -28,900; Buckland *et al.* 1992). These estimates are both more than 15 years old and the data available do not allow us to determine if they remain valid.

Forcada *et al.* (1996) estimated there were 3,583 fin whales in the western Mediterranean (standard error = 967; 95% confidence interval = 2,130 - 6,027), which is similar to an estimate published by Notarbartolo-di-Sciara *et al.* (2003). In the Mediterranean's Ligurian Sea (which includes the Pelagos Whale Sanctuary and the Gulf of Lions), Forcada *et al.* (1995) estimated there were 901 fin whales (standard error = 196.1).

Regardless of which of these estimates, if any, come closest to actual population sizes, these estimates suggest that the global population of fin whales consists of tens of thousands of individuals. Based on ecological theory and demographic patterns derived from several hundred imperiled species and populations, fin whales appear to exist at population sizes that are large enough to avoid demographic phenomena that are known to increase the extinction probability of species that exist as "small" populations (that is, “small” populations experience phenomena such as demographic stochasticity, inbreeding depression and Allee effects, among others, that cause their population size to become a threat in and of itself). As a result, we assume that fin whales are likely to be threatened more by exogenous threats such as anthropogenic activities (primarily whaling, entanglement, and ship strikes) or natural phenomena (such as disease, predation, or changes in the distribution and abundance of their prey in response to changing climate) than endogenous threats caused by the small size of their population.

Nevertheless, based on the evidence available, the number of fin whales that are recorded to have been killed or injured in the past 20 years by human activities or natural phenomena, does not appear to be increasing the extinction probability of fin whales, although it may slow the rate at which they recover from population declines that were caused by commercial whaling.

Diving and Social Behavior

The percentage of time fin whales spend at the surface varies. Some authors have reported that fin whales make 5-20 shallow dives with each of these dives lasting 13-20 seconds followed by a deep dive lasting between 1.5 and 15 minutes (Gambell 1985). Other authors have reported that the fin whale’s most common dives last between 2 and 6 minutes, with 2 to 8 blows between dives (Hain *et al.* 1992, Watkins 1981).

In waters off the Atlantic Coast of the U.S. individual fin whales or pairs represented about 75% of the fin whales observed during the Cetacean and Turtle Assessment Program (Hain *et al.* 1992). Individual whales or groups of less than five individuals represented about 90% of the observations (out of 2,065 observations of fin whales, the mean group size was 2.9, the modal value was 1, and the range was 1 – 65 individuals; Hain *et al.* 1992).

Vocalizations and Hearing

The sounds fin whales produce underwater are one of the most studied *Balaenoptera* sounds. Fin whales produce a variety of low-frequency sounds in the 10-200 Hz band (Watkins 1981; Watkins *et al.* 1987a; Edds 1988; Thompson *et al.* 1992). The most typical signals are long, patterned sequences of short duration (0.5-2s) infrasonic pulses in the 18-35 Hz range (Patterson and Hamilton 1964). Estimated source levels are as high as 190 dB (Patterson and Hamilton 1964; Watkins *et al.* 1987a; Thompson *et al.* 1992; McDonald *et al.* 1995). 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 (Clark and Charif 1998). Short sequences of rapid pulses in the 20-70 Hz band are associated with animals in social groups (McDonald *et al.* 1995, Clark personal communication, McDonald personal communication). Each pulse lasts on the order of one second and contains twenty cycles (Tyack 1999).

During the breeding season, fin whales produce a series of pulses in a regularly repeating pattern. These bouts of pulsing may last for longer than one day (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.* (1995) 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.* 1992).

As with other vocalizations produced by baleen whales, the function of fin whale vocalizations is unknown, although there are numerous hypotheses (which include: maintenance of inter-individual distance, species and individual recognition, contextual information transmission, maintenance of social organization, location of topographic features, and location of prey resources; see the review by Thompson *et al.* 1992 for more information on these hypotheses). Responses to conspecific sounds have been demonstrated in a number of mysticetes, and there is no reason to believe that fin whales do not communicate similarly (Edds-Walton 1997). The low-frequency sounds produced by fin whales have the potential to travel over long distances, and it is possible that long-distance communication occurs in fin whales (Payne and Webb 1971; Edds-Walton 1997). Also, there is speculation that the sounds may function for long-range echolocation of large-scale geographic targets such as seamounts, which might be used for orientation and navigation (Tyack 1999).

Humpback Whale

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 reproduce and give birth to calves) and cooler, temperate or sub-Arctic waters in summer months (where they feed). In their summer foraging areas and winter calving areas, humpback whales tend to occupy shallower, coastal waters; during their seasonal migrations, however, humpback whales disperse widely in deep, pelagic waters and tend to avoid shallower coastal waters (Winn and Reichley 1985).

In the North Pacific Ocean, the summer range of humpback whales includes coastal and inland waters from Point Conception, California, north to the Gulf of Alaska and the Bering Sea, and west along the Aleutian Islands to the Kamchatka Peninsula and into the Sea of Okhotsk (Tomlin 1967, Nemoto 1957, Johnson and Wolman 1984 as cited in NMFS 1991b). These whales migrate to Hawaii, southern Japan, the Mariana Islands, and Mexico during the winter.

In the Atlantic Ocean, humpback whales range from the mid-Atlantic bight, the Gulf of Maine, across the southern coast of Greenland and Iceland, and along coast of Norway in the Barents Sea. These humpback whales migrate to the western coast of Africa and the Caribbean Sea during the winter.

In the Southern Ocean, humpback whales occur in waters off Antarctica. These whales migrate to the waters off Venezuela, Brazil, southern Africa, western and eastern Australia, New Zealand, and islands in the southwest Pacific during the austral winter. A separate population of humpback whales appears to reside in the Arabian Sea in the Indian Ocean off the coasts of Oman, Pakistan, and India (Mikhalev 1997).

Population Structure

Descriptions of the population structure of humpback whales differ depending on whether an author focuses on where humpback whales winter or where they feed. During winter months in northern or southern hemispheres, adult humpback whales migrate to specific areas in warmer, tropical waters to reproduce and give birth to calves. During summer months, humpback whales migrate to specific areas in northern temperate or sub-arctic waters to forage. In summer months, humpback whales from different "reproductive areas" will congregate to feed; in the winter months, whales will migrate from different foraging areas to a single wintering area. In either case, humpback whales appear to form "open" populations; that is, populations that are connected through the movement of individual animals.

NMFS' Stock Assessment Reports recognize four stocks of humpback whales in the North Pacific Ocean, based on genetic and photo-identification studies: two Eastern North Pacific stocks, one Central North Pacific stock, and one Western Pacific stock (Hill and DeMaster 1998). The first two of these stocks are based on where these humpback whales winter: the central North Pacific stock winters in the waters around Hawai'i while the eastern North Pacific stock (also called the California-Oregon-Washington-Mexico stock) winters along coasts of Central America and Mexico.

Herman (1979) presented extensive evidence and various lines of reasoning to conclude that the humpback whales associated with the main Hawaiian Islands immigrated to those waters only in the past 200 years. Winn and Reichley (1985) identified genetic exchange between the humpback whales that winter off Hawai'i and those that winter off Mexico (with further mixing on feeding areas in Alaska) and suggested that the humpback whales that winter in Hawai'i may have emigrated from wintering areas in Mexico. Based on these patterns of movement, we conclude that the various stocks of humpback whales are not true populations or, at least, they represent populations that experience substantial levels of immigration and emigration.

In the Atlantic Ocean, humpback whales aggregate in four feeding areas in the summer months: (1) Gulf of Maine, eastern Canada, (2) west Greenland, (3) Iceland and (4) Norway (Katona and Beard 1990, Smith *et al.* 1999). The principal breeding range for these whales lies from the Antilles and northern Venezuela to Cuba (Winn *et al.* 1975, Balcomb and Nichols 1982, Whitehead and Moore 1982). The largest contemporary breeding aggregations occur off the Greater Antilles where humpback whales from all of the North Atlantic feeding areas have been identified from photographs (Katona and Beard 1990, Clapham *et al.* 1993b, Mattila *et al.* 1994, Palsbøll *et al.* 1997, Smith *et al.* 1999, Stevick *et al.* 2003a). Historically, an important breeding aggregation was located in the eastern Caribbean based on the important humpback whale fisheries this region supported (Mitchell and Reeves 1983, Reeves *et al.* 2001, Smith and Reeves 2003). Although sightings persist in those areas, modern humpback whale abundance appears to be low (Winn *et al.* 1975, Levenson and Leapley 1978, Swartz *et al.* 2003). Winter aggregations also occur at the Cape Verde Islands in the Eastern North Atlantic (Reiner *et al.* 1996, Reeves *et al.* 2002, Moore *et al.* 2003). In another example of the "open" structure of humpback whale populations, an individual humpback whale migrated from the Indian Ocean to the South Atlantic Ocean and demonstrated that individual whales may migrate from one ocean basin to another (Pomilla and Rosenbaum 2005).

A separate population of humpback whales appears to reside in the Arabian Sea in the Indian Ocean off the coasts of Oman, Pakistan, and India (Mikhalev 1997).

Threats to the Species

Natural Threats. There is limited information on natural phenomena that kill or injure humpback whales. We know that humpback whales are killed by orcas (Dolphin 1989, Florez-González *et al.* 1984, Whitehead and Glass 1985) and are probably killed by false killer whales and sharks. Because 7 female and 7 male humpback whales stranded on the beaches of Cape Cod and had died from toxin produced by dinoflagellates between November 1987 and January 1988, we also know that adult and juvenile humpback whales are killed by naturally-produced biotoxins (Geraci *et al.* 1989).

Other natural sources of mortality, however, remain largely unknown. Similarly, we do not know whether and to what degree natural mortality limits or restricts patterns of growth or variability in humpback whale populations.

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 humpback whales and was ultimately responsible for listing humpback whales as an endangered species. From 1900 to 1965, nearly 30,000 whales were taken in modern whaling operations of the Pacific Ocean. Prior to that, an unknown number of humpback whales were taken (Perry *et al.* 1999). In 1965, the International Whaling Commission banned commercial hunting of humpback whales in the Pacific Ocean. As its legacy, whaling has reduced humpback whales to a fraction of their historic population size and, as a result, makes it easier for other human activities to push these whales closer to extinction.

Humpback whales are also killed or injured during interactions with commercial fishing gear. Like fin whales, humpback whales have been entangled by fishing gear off Newfoundland and Labrador, Canada: a total of 595 humpback whales were reported to have been captured in coastal fisheries in those two provinces between 1969 and 1990 (Lien 1994, Perkins and Beamish 1979). Of these whales, 94 are known to have died as a result of that capture, although, like fin whales, most of the animals that died were smaller: less than 12 meters in length (Lien 1994). These data suggest that, despite their size and strength, humpback whales are likely to be entangled and, in some cases, killed by gear used in modern fisheries.

Along the Atlantic Coast of the U.S. and the Maritime Provinces of Canada, there were 101 confirmed entanglement events between 2006 and 2010 (Henry et al. 2012). Of these, 20 resulted in serious injury and 9 resulted in mortality of humpbacks. No information is available on the number of humpback whales that have been killed or seriously injured by interactions with fishing fleets outside of U.S. waters.

There are also reports of entangled humpback whales from the Hawaiian Islands. From 2001 through 2006, there were 23 reports of entangled humpback whales in Hawaiian waters; 16 of these reports were from 2005 and 2006. Some of the entangled humpback whales observed in Hawaiian waters brought the gear with them from higher latitude feeding grounds.

The number of humpback whales killed by ship strikes is exceeded only by fin whales. As of 2008, there have been more than 143 recorded ship strikes involving humpback whales worldwide (Van Waerebeek and Leaper 2008), and many likely go undetected or unreported. When cause of death could be determined, ship strikes made up 4% of observed humpback whale deaths in 2001-2005 (Nelson *et al.* 2007) and 2003-2007 (Glass *et al.* 2009) along the US east coast, Canadian Maritimes and Gulf of Mexico region. On the Pacific coast, the average number of documented humpback whale deaths by ship strikes for 2004-2008 is 0.4 per year (Carretta et al. 2012). While there are reports of humpback whales being injured as a result of ship strikes outside of U.S. waters, we do not have more specific data.

Status

Humpback whales were listed as endangered under the ESA in 1973. Humpback whales are listed as endangered on the IUCN Red List of Threatened Animals (Baillie and Groombridge 1996). They are also protected by the Convention on International Trade in Endangered Species of wild flora and fauna and the MMPA. Critical habitat has not been designated for humpback whales.

It is difficult to assess the current status of humpback whales for the same reasons that it is difficult to assess the status of fin whales: (1) there is no general agreement on the size of the humpback whale population prior to whaling and (2) estimates of the current size of the different humpback whale populations vary widely and produce estimates that are not always comparable to one another, although robust estimates of humpback whale populations in the western North Atlantic have been published. We may never know the size of the humpback whale population prior to whaling.

Winn and Reichley (1985) argued that the global population of humpback whales consisted of at least 150,000 whales in the early 1900s, with the largest population historically occurring in the Southern Ocean. Based on analyses of mutation rates and estimates of genetic diversity, Palumbi and Roman (2006) concluded that there may have been as many as 240,000 (95% confidence interval = 156,000 – 401,000) humpback whales in the North Atlantic before whaling began. In the western North Atlantic between Davis Strait, Iceland and the West Indies, Mitchell and Reeves (1983) estimated there were at least 4,685 humpback whales in 1865 based on available whaling records (although the authors note that this does not represent a "pre-exploitation estimate" because whalers from Greenland, the Gulf of St. Lawrence, New England, and the Caribbean Sea had been hunting humpback whales before 1865).

Estimates of the number of humpback whales occurring in the different populations that inhabit the Northern Pacific population have risen over time. In the 1980s, the size of the North Pacific humpback whale population was estimated to range from 1,407 to 2,100 (Baker 1985; Darling and Morowitz 1986; Baker and Herman 1987). By the mid-1990s, the population was estimated to consist of about 6,000 whales (standard error = 474) in the North Pacific (Calambokidis *et al.* 1997; Cerchio 1998; Mobley *et al.* 1999).

Between 2004 and 2006, an international group of whale researchers coordinated their surveys to conduct a comprehensive assessment of the population structure, levels of abundance, and status of humpback whales in the North Pacific (Calambokidis *et al.* 2008). That effort identified a total of 7,971 unique individuals from photographs taken during close approaches. Of this total, 4,516 individuals were identified at wintering regions in at least one of the three seasons in which the study surveyed wintering areas and 4,328 individuals were identified at least once at feeding areas in one of the two years in which the study surveyed feeding areas. Based on the results of that effort, Calambokidis *et al.* (2008) estimated that the population of humpback whales in the North Pacific Ocean consisted of about 18,300 whales, not counting calves. Almost half of the humpback whales that were estimated to occur in wintering areas, or about 8,000 humpback whales, occupy the Hawaiian Islands during the winter months.

In the North Atlantic, Stevick *et al.* (2003) estimated the size of the humpback whale population between 1979 and 1993 by applying statistical analyses that are commonly used in capture-recapture studies to individual humpback whales that were identified based on natural markings. Between 1979 and 1993, they estimated that the North Atlantic populations (what they call the "West Indies breeding population") consisted of between 5,930 and 12,580 individual whales. The best estimate they produced (11,570; 95% confidence interval = 10,290 -13,390) was based on samples from 1992 and 1993. If we assume that this population has grown according to the instantaneous rate of increase Stevick *et al.* (2003) estimated for this population ($r = 0.0311$), this would lead us to estimate that this population could be over 20,000 individuals in 2011-2012.

Regardless of which of these estimates, if any, most closely correspond to the actual size and trend of the humpback whale population, all of these estimates suggest that the global population of humpback whales consists of tens of thousands of individuals. Based on ecological theory and demographic patterns derived from several hundred imperiled species and populations, humpback whales appear to exist at population sizes that are large enough to avoid demographic phenomena that are known to increase the extinction probability of species that exist as “small” populations (that is, “small” populations experience phenomena such as demographic stochasticity, inbreeding depression, and Allee effects, among others, that cause their population size to become a threat in and of itself). As a result, we assume that humpback whales will have elevated extinction probabilities because of exogenous threats caused by anthropogenic activities (primarily whaling, entanglement, and ship strikes) and natural phenomena (such as disease, predation, or changes in the distribution and abundance of their prey in response to changing climate) rather than endogenous threats caused by the small size of their population.

Diving and Social Behavior

In Hawaiian waters, humpback whales remain almost exclusively within the 1820 m isobath and usually within waters depths less than 182 meters. Maximum diving depths are approximately 150 m (492 ft) (but usually <60 m [197 ft]), with a very deep dive (240 m [787 ft]) recorded off Bermuda (Hamilton *et al.* 1997). They may remain submerged for up to 21 min (Dolphin 1987). Dives on feeding grounds ranged from 2.1-5.1 min in the North Atlantic (Goodyear unpublished manuscript). In southeast Alaska average dive times were 2.8 min for feeding whales, 3.0 min for non-feeding whales, and 4.3 min for resting whales (Dolphin 1987). In the Gulf of California humpback whale dive times averaged 3.5 min (Strong 1989). Because most humpback prey is likely found above 300 m depths most humpback dives are probably relatively shallow.

In a review of the social behavior of humpback whales, Clapham (1986) reported that they form small, unstable social groups during the breeding season. During the feeding season they form small groups that occasionally aggregate on concentrations of food. Feeding groups are sometimes stable for long-periods of times. There is good evidence of some territoriality on feeding (Clapham 1994, 1996), and calving areas (Tyack 1981). 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 polygyny (Clapham 1996). Intermale competition for proximity to females can be intense as expected by the sex ratio on the breeding grounds which may be as high as 2.4:1.

Vocalizations and Hearing

Humpback whales produce at least three kinds of vocalization: (1) complex songs with components ranging from at least 20Hz to 4 kHz with estimated source levels from 144dB 174 dB, which are mostly produced by males on breeding areas (Payne 1970, Winn *et al.* 1970, Richardson *et al.* 1995); (2) social sounds in breeding areas that extend from 50 Hz B more than 10 kHz with most energy below 3 kHz

(Tyack and Whitehead 1983, Richardson *et al.* 1995); and (3) vocalizations in foraging areas that are less frequent, but tend to be 20Hz to 2 kHz with estimated sources levels in excess of 175 dB re 1 µPa-m (Thompson *et al.* 1986, Richardson *et al.* 1995). Sounds that investigators associate with aggressive behavior in male humpback whales are very different from songs; they extend from 50 Hz to 10 kHz (or higher), with most energy in components below 3 kHz (Tyack 1983, Silber 1986). These sounds appear to have an effective range of up to 9 kilometers (Tyack and Whitehead 1983). A general description of the anatomy of the ear for cetaceans is provided in the description of the blue whale above; that description is also applicable to humpback whales.

In summary, humpback whales produce at least three kinds of sounds:

> a. Complex songs with components ranging from at least 20 Hz–4 kHz with estimated source levels from 144 – 174 dB; these are mostly sung by males on the breeding grounds (Au et al 2006; Frazer and Mercado 2000; U.S. Navy 2006a; Payne 1970; Winn *et al.* 1970a; Richardson *et al.* 1995)
>
> b. Social sounds in the breeding areas that extend from 50Hz–more than 10 kHz with most energy below 3 kHz (Tyack and Whitehead 1983, Richardson *et al.* 1995); and
>
> c. Feeding area vocalizations that are less frequent, but tend to be 20 Hz–2 kHz with estimated sources levels in excess of 175 dB re 1 µPa-m (Thompson *et al.* 1986; Richardson *et al.* 1995).

Helwig *et al.* (2000) produced a mathematical model of a humpback whale's hearing sensitivity based on the anatomy of the whale's ear. Based on that model, they concluded that humpback whales would be sensitive to sound in frequencies ranging from 0.7 kHz to 10 kHz, with a maximum sensitivity between 2 and 6 kHz.

North Atlantic Right Whale

Distribution

Right whales exist as three separate species: North Atlantic right whales (*Eubalaena glacialis*) that are distributed seasonally from the Gulf of Mexico north to waters off Newfoundland and Labrador (on the western Atlantic) and from northern Africa and Spain north to waters north of Scotland and Ireland (the Shetland and Orkney Islands; on the eastern Atlantic coast); North Pacific right whales (*E. japonica*) that historically ranged seasonally from the coast of Baja California north to the northern Bering Sea (on the eastern Pacific) and the south China Sea north to the Sea of Okhotsk and the Kamchatka Peninsula (on the western Pacific); and Southern right whales (*E. australis*) which historically ranged across the Southern

Ocean, including waters off southern Australia, New Zealand, Chile, Argentina, and southern Africa (north to Madagascar). This Opinion will only discuss the North Atlantic right whale.

In the western Atlantic Ocean, right whales generally occur in northwest Atlantic waters west of the Gulf Stream and are most commonly associated with cooler waters (21 C). North Atlantic right whales are most abundant in Cape Cod Bay between February and April (Hamilton and Mayo 1990 Schevill *et al*. 1986, Watkins and Schevill 1982), in the Great South Channel in May and June (Kenney *et al.* 1986, Payne *et al*. 1990), and off Georgia and Florida from mid-November through March (Slay *et al*. 1996). Right whales also frequent the Bay of Fundy, Browns and Baccaro Banks (in Canadian waters), Stellwagen Bank and Jeffrey's Ledge in the spring and summer months, and use mid-Atlantic waters as a migratory pathway between the winter calving grounds and their spring and summer nursery feeding areas in the Gulf of Maine. North Atlantic right whales are not found in the Caribbean Sea and have been recorded only rarely in the Gulf of Mexico.

Population Structure

NMFS recognizes two extant groups of right whales in the North Atlantic Ocean (*E. glacialis*): an eastern population and a western population. A third population may have existed in the central Atlantic (migrating from east of Greenland to the Azores or Bermuda), but appears to be extinct, if it existed as a distinct population at all (Perry *et al*. 1999).

The degree to which the two extant populations of North Atlantic right whales are connected through immigration or emigration is unknown, but the two populations have historically been treated as if they are isolated populations. Nevertheless, on 5 January 2009, a North Atlantic right whale that had been observed in the Bay of Fundy on 24 September 2008, was observed in the Azore Islands (38 22.698 N and 28 30.341W) which demonstrates that at least one right whale migrated across the Atlantic (Hamilton *et al.* 2009). These long-range sightings indicate an extended range for at least some individuals and perhaps the existence of important habitat areas not presently well described. The few published records from the Gulf of Mexico (Moore and Clark 1963; Schmidly *et al.* 1972) represent either distributional anomalies, normal wanderings of occasional animals, or a more extensive historic range beyond the sole known calving and wintering ground in the waters of the southeastern United States.

However, from 1980 to 2001, there were 64 calves born that were not sighted later with their mothers (Frasier *et al.* 2007). Fathers for only 45% of known calves have been genetically determined. However, genetic profiles were available for 69% of all photo-identified males (Frasier 2005). The conclusion was that the majority of these calves must have different fathers that cannot be accounted for by the unsampled males and that the population of males must be larger than previously thought (Frasier 2005). This inference of additional animals that have never been captured photographically and/or genetically suggests the existence of habitats of potentially significant use that remain unknown and suggests emigration or immigration with other unknown populations.

Threats to the Species

Natural threats. Several researchers have suggested that the recovery of right whales in the northern hemisphere has been impeded by competition with other whales for food (Rice 1974, Scarff 1986). Mitchell (1975) analyzed trophic interactions among baleen whales in the western North Atlantic and noted that the foraging grounds of right whales overlapped with the foraging grounds of sei whales and both preferentially feed on copepods. Reeves *et al.* (1978) noted that several species of whales feed on copepods in the eastern North Pacific, so that the foraging pattern and success of right whales would be affected by other whales as well. Mitchell (1975) argued that the right whale population in the North Atlantic had been depleted by several centuries of whaling before steam-driven boats allowed whalers to hunt sei whales; from this, he hypothesized that the decline of the right whale population made more food available to sei whales and helped their population to grow. He then suggested that competition with the sei whale population impedes or prevents the recovery of the right whale population.

Anthropogenic threats. Several human activities are known to threaten North Atlantic right whales: whaling, commercial fishing and shipping. Historically, whaling represented the greatest threat to every population of right whales and was ultimately responsible for their listing as an endangered species. As its legacy, whaling reduced North Atlantic right whales to about 396 (minimum population as of 2007) individuals in the western North Atlantic Ocean; the North Atlantic right whale population in the eastern North Atlantic Ocean is probably much smaller, although we cannot estimate the size of that population from the data available.

Of the current threats to North Atlantic right whales, entanglement in commercial fishing gear and ship strikes currently pose the greatest threat to the persistence of North Atlantic right whales. Along the Atlantic Coast of the U.S. and the Maritime Provinces of Canada, there were 33 confirmed reports of right whales being entangled in fishing gear between 2006 and 2010 (Henry et al. 2012). Of these, right whales were injured in five of the entanglements and killed in four entanglements.

In the same region, there were 13 confirmed reports of right whales being struck by vessels between 2006 and 2010 (Henry et al. 2012). Of these, right whales were injured in one of the ship strikes and killed in five ship strikes. In April 2009, a research vessel in the Stellwagen Bank National Marine Sanctuary struck a North Atlantic right whale while transiting to port. Pictures of the whale taken minutes after the strike revealed that the propeller had struck and cut the animal. Although the animal was injured the injury was deemed not life threatening.

The rate at which North Atlantic right whales are killed or injured by ship strikes and in entanglements also appears to be somewhat variable, or possibly increasing: from 1999 to 2003, about 2.6 right whales were killed per year; from 2001 to 2005, an average of 3.2 right whales were killed per year (NMFS 2005, Waring *et al*. 2007). The most recent estimate of anthropogenic mortality and serious injury available shows a rate of 3.0 right whales per year from 2006 to 2010. Of these, 1.2 were attributed to ship strikes and 1.8 were attributed to entanglements (Henry *et al*. 2012).

Status

Right whales (*E. glacialis* and *E. australi*s) were listed as endangered under the ESA in 1970. In April, 2008, NMFS divided northern right whales into two separate listings: North Atlantic right whales (*E. glacialis*) and North Pacific right whales (*E. japonica*). All three species (*E. glacialis*, *E. japonica,* and Southern right whales, *E. australis*) are listed as endangered. Since 1949, the northern right whale has been protected from commercial whaling by the International Whaling Commission. They are also protected by the Convention on International Trade in Endangered Species of wild flora and fauna and the MMPA. NMFS designated critical habitat for the North Atlantic population of right whales on 3 June 1994 (59 FR 28793), and this remains in effect for the North Atlantic right whale.

The legacy effects of whaling appear to have had and continue to have greatest effect on endangered North Atlantic right whales by reducing them to a population size that is sufficiently small to experience “small population dynamics” (Caughley 1994, Lande 1993, Lande *et al*. 2003, Melbourne and Hastings 2008). Kraus *et al*. (2005) estimated that about 350 individual right whales, including about 70 mature females, occur in the western North Atlantic. The western North Atlantic population numbered at least 361 individuals in 2005 and at least 396 in 2007 (Waring et al. 2012). Examination of the minimum number alive population index calculated from the individual sightings database, as it existed on 6 July 2010, for the years 1990-2007, suggests a positive trend (2.4 % mean growth rate) in population size. These data reveal a significant increase in the number of catalogued whales alive during this period, but with significant variation due to apparent losses (particularly pregnant females and calves) exceeding gains during 1998-99 (Waring et al. 2012).

Nevertheless, at these population sizes, we would expect North Atlantic right whales to have higher probabilities of becoming extinct because of demographic stochasticity, demographic heterogeneity (Coulson *et al*. 2006, Fox *et al*. 2006) —including stochastic sex determination (Lande *et al*. 2003) — and the effects of these phenomena interacting with environmental variability. Demographic stochasticity refers to the randomness in the birth or death of an individual in a population, which results in random variation on how many young that individuals produce during their lifetime and when they die. Demographic heterogeneity refers to variation in lifetime reproductive success of individuals in a population (generally, the number of reproductive adults an individual produces over their reproductive lifespan), such that the deaths of different individuals have different effects on the growth or decline of a population (Coulson *et al*. 2006). Stochastic sex determination refers to the randomness in the sex of offspring such that sexual ratios in population fluctuate over time (Melbourne and Hastings 2008).

At small population sizes, populations experience higher extinction probabilities because of their population size, because stochastic sexual determination can leave them with all males or all females (which occurred to the heath hen and dusky seaside sparrow just before they became extinct), or because the loss of individuals with high reproductive success has a disproportionate effect on the rate at which the population declines (Coulson *et al*. 2006). In general, an individual's contribution to the growth (or decline) of the population it represents depends, in part, on the number of individuals in the population: the smaller the population, the more the performance of a single individual is likely to affect the population's growth or decline (Coulson *et al*. 2006). Given the small size of the northern right whale population, the performance ("fitness" measured as the longevity of individuals and their reproductive success over their lifespan) of individual whales would be expected to have appreciable consequences for the growth or decline of the northern right whale population. Evidence of the small population dynamics of North Atlantic right whales appears in demographic models that suggest that the death or survival of one or two individual animals is sufficient to determine whether North Atlantic right whales are likely to accelerate or abate the rate at which their population continues to decline (Fujiwara and Caswell 2001).

These phenomena would increase the extinction probability of northern right whales and amplify the potential consequences of human-related activities on this species. Based on their population size and population ecology (that is, slow-growing mammals that give birth to single calves with several years between births), we assume that right whales would have elevated extinction probabilities because of exogenous threats caused by anthropogenic activities that result in the death or injury of individual whales (for example, ship strikes or entanglement) and natural phenomena (such as disease, predation, or changes in the distribution and abundance of their prey in response to changing climate) *as well as* endogenous threats resulting from the small size of their population. Based on the number of other species in similar circumstances that have become extinct (and the small number of species that have avoided extinction in similar circumstances), the longer North Atlantic right whales remain in these circumstances, the greater their extinction probability becomes.

Diving and Social Behavior

Right whales dive as deep as 306 m (Mate *et al.* 1992). In the Great South Channel, average diving time is close to 2 minutes; average dive depth is 7.3 m with a maximum of 85.3 m (Winn *et al.* 1994). In the U.S. Outer Continental Shelf the average diving time is about 7 min although maximum dive durations are considerably longer (CeTAP 1982). For example, Baumgartner and Mate (2003) reported right whale feeding dives were characterized by a rapid descent from the surface to a particular depth between 80 and 175 m (262 to 574 ft) with animals remaining at those depths for 5 to 14 min, then ascending quickly to the surface (Baumgartner and Mate 2003). Longer surface intervals have been observed for reproductively active females and their calves (Baumgartner and Mate, 2003).

Northern right whales are primarily seen in groups of less than 12, most often singles or pairs (Jefferson *et al.* 1993). They may form larger groups while on feeding or breeding areas (Jefferson *et al.* 1993).

Vocalizations and Hearing

North Atlantic right whales produce a variety of sounds, including moans, screams, gunshots, blows, upcalls, downcalls, and warbles that are often linked to specific behaviors (Matthews et al., 2001; Laurinolli et al., 2003; Vanderlaan et al., 2003; Parks et al., 2005; Parks and Tyack, 2005). Sounds can be divided into three main categories: (1) blow sounds; (2) broadband impulsive sounds; and (3) tonal call types (Parks and Clark, 2007). Blow sounds are those coinciding with an exhalation; it is not known whether these are intentional communication signals or just produced incidentally (Parks and Clark, 2007).

Broadband sounds include non-vocal slaps (when the whale strikes the surface of the water with parts of its body) and the “gunshot” sound; data suggests that the latter serves a communicative purpose (Parks and Clark, 2007). Tonal calls can be divided into simple, low-frequency, stereo-typed calls and more complex, frequency-modulated, higher-frequency calls (Parks and Clark 2007). Most of these sounds range in frequency from 0.02 to 15 kHz (dominant frequency range from 0.02 to less than 2 kHz; durations typically range from 0.01 to multiple seconds) with some sounds having multiple harmonics (Parks and Tyack 2005).

Source levels for some of these sounds have been measured as ranging from 137 to 192 dB root-mean-square (rms) re 1 Pa-m (decibels at the reference level of one micro Pascal at one meter) (Parks *et al.*, 2005; Parks and Tyack, 2005). Parks and Clark (2005) suggested that the frequency of right whale vocalizations increases significantly during the period from dusk until dawn. Recent morphometric analyses of North Atlantic right whale inner ears estimates a hearing range of approximately 0.01 to 22 kHz based on established marine mammal models (Parks *et a*l. 2004, Parks and Tyack 2005, Parks *et al*. 2007). In addition, Parks *et al*. (2007) estimated the functional hearing range for right whales to be 15 Hz to 18 kHz.

Designated Critical Habitat

Five areas have been reported to be critical to the survival and recovery of North Atlantic right whales: (1) coastal Florida and Georgia; (2) the Great South Channel, which lies east of Cape Cod; (3) Cape Cod and Massachusetts Bays; (4) the Bay of Fundy; and (5) Browns and Baccaro Banks off southern Nova Scotia. The first three areas occur in U.S. waters and have been designated by NMFS as critical habitat (59 FR 28793). North Atlantic right whales are most abundant in Cape Cod Bay between February and April (Hamilton and Mayo 1990; Schevill *et al*. 1986; Watkins and Schevill 1982), in the Great South Channel in May and June (Kenney *et al*. 1986, Payne *et al*. 1990), and off Georgia/Florida from mid-November through March (Slay *et al*. 1996). Right whales also frequent the Bay of Fundy, Browns and Baccaro Banks (in Canadian waters), Stellwagen Bank and Jeffrey's Ledge in spring and summer months and use mid-Atlantic waters as a migratory pathway between winter calving grounds and their spring and summer nursery/feeding areas in the Gulf of Maine. A recent review and comparison of sighting data suggests that Jeffrey's Ledge may also be regularly used by right whales in late fall (October through December; Weinrich *et al*. 2000).

The availability of dense concentrations of zooplankton blooms in Cape Cod Bay in late winter and the Great South Channel in spring is described as the key factor for right whale utilization of these areas. Kraus and Kenney (1991) provide an overview of data regarding right whale use of these areas. Important habitat components in Cape Cod Bay include seasonal availability of dense zooplankton patches and protection from weather afforded by land masses surrounding the bay. The spring current regime and bottom topography of the Great South Channel result in nutrient rich upwelling conditions. These conditions support the dense plankton and zooplankton blooms utilized by right whales. The combination of highly oxygenated water and dense zooplankton concentrations are optimal conditions for the small schooling fishes (sand lance, herring and mackerel) that prey upon some of the same zooplankton as right whales. Therefore, the abundance of these fishes, in turn, may affect and be affected by the distribution of several piscivorous marine mammal species such as humpback, fin, minke, and pilot whales, Atlantic whitesided dolphins, and harbor porpoise (CeTAP 1982).

Overfishing has severely reduced the stocks of several groundfish species such as cod, haddock, and yellowtail flounder. Recovery of commercially targeted finfish stocks from their current overfished condition may reduce the biomass of small schooling fish that feed directly on zooplankton resources throughout the region. It is unknown whether zooplankton densities that occur seasonally in Cape Cod Bay or the Great South Channel could be expected to increase significantly. However, increased predation by groundfish on small schooling fish in certain areas and at specific critical periods may allow the necessary high zooplankton densities to be maintained in these areas for longer periods, or accumulate in other areas at levels acceptable to right whales.

Fishing is allowed within the Cape Cod Bay and Great South Channel right whale critical habitat. Lobster trap gear and anchored gillnet gear are believed to pose the most serious risks of entanglement and serious injury to right whales frequenting these waters. As a result, regulations developed under the Atlantic Large Whale Take Reduction Program restrict the use of lobster and anchored gillnet gear in Cape Cod Bay and Great South Channel critical habitat. The most restrictive measures apply during peak right whale abundance: January 1 to May 15 in Cape Cod Bay, and April 1 to June 30 in the Great South Channel critical habitat. Measures include prohibitions on the use of lobster trap gear and anchored gillnet gear in the Great South Channel critical habitat during periods of peak right whale abundance (with the exception of gillnet gear in the Great South Channel Sliver Area), and, for Cape Cod Bay critical habitat, anchored gillnet gear prohibitions and lobster trap restrictions during peak right whale abundance. During non-peak periods of right whale abundance, lobster trap and gillnet fishers must modify their gear by using weak links in net and/or buoy lines, follow gillnet anchoring requirements and meet mandatory breaking strengths for buoy line weak links, amongst others. Additional measures (i.e., gear marking requirements, and prohibitions on the use of floating line and the wet storage of gear) apply within as well as outside of critical habitat. All of these measures are intended to reduce the likelihood of whale entanglements or the severity of an entanglement should an animal encounter anchored gillnet or lobster gear.

The critical habitat identified in the Southeast U.S. is used primarily as a calving and nursery area. The nearshore waters of northeast Florida and southern Georgia were formally designated as critical habitat for right whales on June 3, 1994 (59 FR 28793); ten years after they were first identified as a likely calving and nursery area for right whales. Since that time, 74 percent of all known, mature female North Atlantic right whales have been documented in this area (Kraus et al., 1993). While sightings off Georgia and Florida include primarily adult females and calves, juveniles and adult males have also been observed.

Sei Whale

Distribution

Sei whales occur in every ocean except the Arctic Ocean. 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 the continental shelf edge (Hain *et al.* 1985); however, this general offshore pattern of sei whale distribution is disrupted during occasional incursions into more shallow and inshore waters (Waring *et al.* 2004).

In the western Atlantic Ocean, sei whales occur from Labrador, Nova Scotia, and Labrador in the summer months and migrate south to Florida and the northern Caribbean (Gambell 1985, Mead 1977). In the eastern Atlantic Ocean, sei whales occur in the Norwegian Sea (as far north as Finnmark in northeastern Norway), occasionally occurring as far north as Spitsbergen Island, and migrate south to Spain, Portugal, and northwest Africa (Jonsgård and Darling 1974, Gambell 1985).

In the North Pacific Ocean, sei whales occur from the Bering Sea south to California (on the east) and the coasts of Japan and Korea (on the west). During the winter, sei whales are found from 20°N to 23°N (Masaki 1977; Gambell 1985). Horwood (1987) reported that 75 - 85% of the North Pacific population of sei whales resides east of 180° longitude.

Sei whales occur throughout the Southern Ocean during the summer months, although they do not migrate as far south to feed as blue or fin whales. During the austral winter, sei whales occur off Brazil and the western and eastern coasts of Southern Africa and Australia.

Population Structure

The population structure of sei whales is largely unknown because there are so few data on this species. The International Whaling Commission’s Scientific Committee groups all of the sei whales in the entire North Pacific Ocean into one population (Donovan 1991). However, some mark-recapture, catch distribution, and morphological research suggest more than one “stock” of sei whales may exist in the Pacific: one between 175^{o}W and 155^{o}W longitude, and another east of 155^{o}W longitude (Masaki 1977); however, the amount of movement between these “stocks” suggests that they probably do not represent demographically-isolated populations as we use this concept in this Opinion.

Mitchell and Chapman (1977) divided sei whales in the western North Atlantic in two populations, one that occupies the Nova Scotian Shelf and a second that occupies the Labrador Sea. Sei whales are most common on Georges Bank and into the Gulf of Maine and the Bay of Fundy during spring and summer, primarily in deeper waters. There are occasional influxes of sei whales further into Gulf of Maine waters, presumably in conjunction with years of high copepod abundance inshore. Sei whales are occasionally seen feeding in association with right whales in the southern Gulf of Maine and in the Bay of Fundy.

Threats to the Species

Natural threats. Sei whales appear to compete with blue, fin, and right whales for prey and that competition may limit the total abundance of each of the species (Rice 1974, Scarff 1986). As discussed previously in the narratives for fin and right whales, the foraging areas of right and sei whales in the western North Atlantic Ocean overlap and both whales feed preferentially on copepods (Mitchell 1975).

Anthropogenic threats. Two human activities are known to threaten sei whales: whaling and shipping. Historically, whaling represented the greatest threat to every population of sei whales and was ultimately responsible for listing sei whales as an endangered species. From 1910 to 1975, approximately 74,215 sei whales were caught in the entire North Pacific Ocean (Horwood 1987, Perry *et al.* 1999). From the early 1900s, Japanese whaling operations consisted of a large proportion of sei whales: 300 - 600 sei whales were killed per year from 1911 to 1955. The sei whale catch peaked in 1959, when 1,340 sei whales were killed. In 1971, after a decade of high sei whale catch numbers, sei whales were scarce in Japanese waters.

In the North Atlantic Ocean, sei whales were hunted from land stations in Norway and Iceland in the early- to mid-1880s, when blue whales started to become more scarce. In the late 1890s, whalers began hunting sei whales in Davis Strait and off the coasts of Newfoundland. In the early 1900s, whalers from land stations on the Outer Hebrides and Shetland Islands started to hunt sei whales. Between 1966 and 1972, whalers from land stations on the east coast of Nova Scotia engaged in extensive hunts of sei whales on the Nova Scotia shelf, killing about 825 sei whales (Mitchell and Chapman 1977).

Sei whales are occasionally killed in collisions with vessels. Of 3 sei whales that stranded along the Atlantic Coast of the U.S. between 1975 and 1996, 2 showed evidence of collisions with ships (Laist *et al.* 2001). Between 1999 and 2010, there were 6 reports of sei whales being struck by vessels along the Atlantic Coast of the U.S. and the Maritime Provinces of Canada (Cole *et al.* 2005, Nelson *et al.* 2007, Henry et al. 2012). Five of these ship strikes were reported as having resulted in the death of the sei whale.

Sei whales are occasionally found entangled. Along the Atlantic Coast of the U.S. and the Maritime Provinces of Canada, there were 3 confirmed reports of sei whales being entangled in fishing gear between 2006 and 2010 (Henry et al. 2012). Of these, sei whales were injured in two of the entanglements and killed in one.

Status

Sei whales were listed as endangered under the ESA in 1973. In the North Pacific, the International Whaling Commission began management of commercial taking of sei whales in 1970, and sei whales were given full protection in 1985 (Allen 1980). Sei whales are also protected by the Convention on International Trade in Endangered Species of wild flora and fauna and the Marine Mammal Protection Act. They are listed as endangered under the IUCN Red List of Threatened Animals (Baillie and Groombridge 1996). Critical habitat has not been designated for sei whales.

Prior to commercial whaling, sei whales in the North Pacific are estimated to have numbered 42,000 individuals (Tillman 1977), although Ohsumi and Fukuda (1975) estimated that sei whales in the North Pacific numbered about 49,000 whales in 1963, had been reduced to 37,000 or 38,000 whales by 1967, and reduced again to 20,600 to 23,700 whales by 1973. When commercial whaling for sei whales ended in 1974, the population of sei whales in the North Pacific had been reduced to between 7,260 and 12,620 animals (Tillman 1977). In the same year, the North Atlantic population of sei whales was estimated to number about 2,078 individuals, including 965 whales in the Labrador Sea group and 870 whales in the Nova Scotia group (IWC 1977, Mitchell and Chapman 1977).

The 2004 abundance estimate of 386 is considered the best available for the Nova Scotia stock of sei whales (Waring et al. 2012). There have been no direct estimates of sei whale abundance in the entire (or eastern) North Pacific based on sighting surveys, but two abundance estimates based on recent line transect surveys of California, Oregon, and Washington waters are 74 (CV=0.88) and 215 (CV=0.71) sei whales, respectively (Forney 2007, Barlow 2010)

Like blue whales, the information available on the status and trend of sei whales do not allow us to reach any conclusions about the extinction risks facing sei whales as a species, or particular populations of sei whales. With the limited data available on sei whales, we do not know whether these whales exist at population sizes large enough to avoid demographic phenomena that are known to increase the extinction probability of species that exist as “small” populations (that is, “small” populations experience phenomena such as demographic stochasticity, inbreeding depression and Allee effects, among others, that cause their population size to become a threat in and of itself) or if sei whales are threatened more by exogenous threats such as anthropogenic activities (primarily whaling, entanglement, and ship strikes) or natural phenomena (such as disease, predation, or changes in the distribution and abundance of their prey in response to changing climate). However, sei whales have historically exhibited sudden increases in abundance in particular areas followed by sudden decreases in number. Several authors have reported “invasion years” in which large numbers of sei whales appeared off areas like Norway and Scotland, followed the next year by sudden decreases in population numbers (Jonsgård and Darling 1974).

With the evidence available, we do not know if this year-to-year variation still occurs in sei whales. However, if sei whales exist as a fraction of their historic population sizes, large amounts of variation in their abundance would increase the extinction probabilities of individual populations (Fagan and Holmes 2006, Fagan *et al.* 1999, 2001).

Diving and Social Behavior

Generally, sei whales make 5-20 shallow dives of 20-30 sec duration followed by a deep dive of up to 15 min (Gambell 1985). The depths of sei whale dives have not been studied, however the composition of their diet suggests that they do not perform dives in excess of 300 meters. Sei whales are usually found in small groups of up to 6 individuals, but they commonly form larger groupings when they are on feeding grounds (Gambell 1985).

Vocalizations and Hearing

There is a limited amount of information on the vocal behavior of sei whales. McDonald et al. (2005) recorded sei whale vocalizations off the Antarctic Peninsula that included broadband sounds in the 100-600 Hz range with 1.5 second duration and tonal and upsweep call in the 200-600 Hz range 1-3 second duration. During visual and acoustic surveys conducted in the Hawaiian Islands in 2002, Rankin and Barlow (2007) recorded 107 sei whale vocalizations, which they classified as two variations of low-frequency downswept calls. The first variation consisted of sweeps from 100 Hz to 44 Hz, over 1.0 seconds. The second variation, which was more common (105 out of 107) consisted of low frequency calls which swept from 39 Hz to 21 Hz over 1.3 seconds. These vocalizations are different from sounds attributed to sei whales in the Atlantic and Southern Oceans but are similar to sounds that had previously been attributed to fin whales in Hawaiian waters. Vocalizations from the North Atlantic consisted of paired sequences (0.5-0.8 sec, separated by 0.4-1.0 sec) of 10-20 short (4 msec) FM sweeps between 1.5-3.5 kHz (Richardson, et al., 1995).

A general description of the anatomy of the ear for cetaceans is provided in the preceding description of the blue whale.

Sperm Whale

Distribution

Sperm whales occur in every ocean except the Arctic Ocean. Sperm whales are found throughout the North Pacific and are distributed broadly from tropical and temperate waters to the Bering Sea as far north as Cape Navarin. Mature, female, and immature sperm whales of both sexes are found in more temperate and tropical waters from the equator to around 45˚ N throughout the year. These groups of adult females and immature sperm whales are rarely found at latitudes higher than 50˚ N and 50˚ S (Reeves and Whitehead 1997). Sexually mature males join these groups throughout the winter. During the summer, mature male sperm whales are thought to move north into the Aleutian Islands, Gulf of Alaska, and the Bering Sea.

In the western Atlantic Ocean, sperm whales are distributed in a distinct seasonal cycle, concentrated east-northeast of Cape Hatteras in winter and shifting northward in spring when whales are found throughout the Mid-Atlantic Bight. Distribution extends further northward to areas north of Georges Bank and the Northeast Channel region in summer and then south of New England in fall, back to the Mid-Atlantic Bight.

In the eastern Atlantic Ocean, mature male sperm whales have been recorded as far north as Spitsbergen (Oien 1990). Recent observations of sperm whales and stranding events involving sperm whales from the eastern North Atlantic suggest that solitary and paired mature male sperm whales predominantly occur in waters off Iceland, the Faroe Islands, and the Norwegian Sea (Gunnlaugsson and Sigurjonsson 1990, Oien 1990, Christensen *et al.* 1992).

In the Mediterranean Sea sperm whales are found from the Alboran Sea to the Levant Basin, mostly over steep slope and deep offshore waters. Sperm whales are rarely sighted in the Sicilian Channel, and are vagrant in the northern Adriatic and Aegean Seas (Notarbartolo di Sciara and Demma 1997). In the Italian seas sperm whales are more frequently associated with the continental slope off western Liguria, western Sardinia, northern and eastern Sicily, and both coasts of Calabria.

Sperm whales have a strong preference for the 3,280 feet (1,000 meters) depth contour and seaward. Berzin (1971) reported that they are restricted to waters deeper than 300 meters (984 feet), while Watkins (1977) and Reeves and Whitehead (1997) reported that they are usually not found in waters less than 1,000 meters (3,281 feet) deep. While deep water is their typical habitat, sperm whales have been observed near Long Island, New York, in water between 41-55 meters (135-180 feet; Scott and Sadove 1997). When they are found relatively close to shore, sperm whales are usually associated with sharp increases in bottom depth where upwelling occurs and biological production is high, implying the presence of good food supplies (Clarke 1956).

Population Structure

The population structure of sperm whales is largely unknown. Lyrholm and Gyllenstein (1998) reported moderate, but statistically significant, differences in sperm whale mitochondrial (mtDNA) between ocean basins, although sperm whales throughout the world appear to be homogenous genetically (Whitehead 2003). Genetic studies also suggest that sperm whales of both genders commonly move across over ocean basins and that males, but not females, often breed in ocean basins that are different from the one in which they were born (Whitehead 2003).

Sperm whales may not form "populations" as that term is normally conceived. Jaquet (1996) outlined a hierarchical social and spatial structure that includes temporary clusters of animals, family units of 10 or 12 females and their young, groups of about 20 animals that remain together for hours or days, "aggregations" and "super-aggregations" of 40 or more whales, and "concentrations" that include 1,000 or more animals (Peterson 1986, Whitehead and Wiegart 1990, Whitehead *et al.* 1991). The "family unit" forms the foundation for sperm whale society and most females probably spend their entire life in the same family unit (Whitehead 2002). The dynamic nature of these relationships and the large spatial areas they are believed to occupy might complicate or preclude attempts to apply traditional population concepts, which tend to rely on group fidelity to geographic distributions that are relatively static over time.

Based on harvests of tagged sperm whales or sperm whales with other distinctive marking, sperm whales in the North Atlantic Ocean appear to represent a single population, with the possible exception of the sperm whales that appear to reside in the Gulf of Mexico.

Female and immature animals stay in Atlantic temperate or tropical waters year round. In the western North Atlantic, groups of female and immature sperm whales concentrate in the Caribbean Sea (Gosho *et al.* 1984) and south of New England in continental-slope and deep-ocean waters along the eastern United States (Blaylock *et al.* 1995). In eastern Atlantic waters, groups of female and immature sperm whales aggregate in waters off the Azores, Madeira, Canary, and Cape Verde Islands (Tomilin 1967). Multiple reports suggest that at least some sperm whales migrate across the North Atlantic Ocean (Mitchell 1975, Martin 1982, Donovan 1991).

The distribution of the sperm whale in the U.S. Exclusive Economic Zone occurs on the continental shelf edge, over the continental slope, and into mid-ocean regions. Waring et al. (1993; Waring et al. 2001) suggest that this offshore distribution is more commonly associated with the Gulf Stream edge and other features. However, the sperm whales that occur in the eastern U.S. Exclusive Economic Zone in the Atlantic Ocean likely represent only a fraction of the total stock. The nature of linkages of the U.S. habitat with those to the south, north, and offshore is unknown. Historical whaling records compiled by Schmidly (1981) suggested an offshore distribution off the southeast United States, over the Blake Plateau, and into deep ocean waters. In the southeast Caribbean, both large and small adults, as well as calves and juveniles of different sizes are reported (Watkins et al. 1985). Whether the northwestern Atlantic population is discrete from northeastern Atlantic is currently unresolved. The International Whaling Commission recognizes one stock for the North Atlantic, based on reviews of many types of stock studies (i.e., tagging, genetics, catch data, mark-recapture, biochemical markers, etc.).

In winter, sperm whales are concentrated east and northeast of Cape Hatteras. In spring, the center of distribution shifts northward to east of Delaware and Virginia and is widespread throughout the central portion of the mid-Atlantic Bight and the southern portion of Georges Bank. In summer, the distribution is similar but now also includes the area east and north of Georges Bank and into the Northeast Channel region, as well as the continental shelf (inshore of the 100-m isobath) south of New England. In the fall, sperm whale occurrence south of New England on the continental shelf is at its highest level, and there remains a continental shelf edge occurrence in the mid-Atlantic Bight. Similar inshore (less than 200 m) observations were made on the southwestern and eastern Scotian Shelf, particularly in the region of "the Gully" (Whitehead and Weilgart 1991).

Several investigators have suggested that the sperm whales that occupy the northern Gulf of Mexico are distinct from sperm whales elsewhere in the North Atlantic Ocean based on year-round presence in the Gulf (Schmidly 1981, Fritts 1983, and Hansen et al. 1996, Mullin and Hoggard 2000). More recent studies provide further support that Gulf of Mexico sperm whales are a separate stock based on year round presence in the Gulf and preliminary results of genetics, size distribution and coda vocalizations (Mullin *et al.* 2003, Jaquet 2006; Jochens *et al.* 2008). These studies, like almost all studies to date, rely almost exclusively on sperm whales from the northern Gulf, with greatest density along and deeper than the 1000 m depth contour, and do not adequately represent sperm whales that may occur regularly in the central, western, southern, or eastern Gulf (for example, Ortega-Ortiz 2003). It is very likely, though, that sperm whales of the north-central Gulf, present there throughout the year (Davis et al. 1998), are more numerous than in other parts of the Gulf (Jochens *et al* 2008). Based on this information NMFS is currently consideringwhether to list the sperm whale population in the northern Gulf of Mexico as a separate species, however, the IWC, does not recognize these sperm whales as a separate stock.

In the Northern Indian Ocean the International Whaling Commission recognized differences between sperm whales in the northern and southern Indian Ocean (Donovan 1991). Little is known about the Northern Indian Ocean population of sperm whales (Perry *et al.* 1999).

Several authors have proposed population structures that recognize at least three sperm whales populations in the North Pacific for management purposes (Kasuya 1991, Bannister and Mitchell 1980). At the same time, the IWC's Scientific Committee designated two sperm whale stocks in the North Pacific: a western and eastern stock or population (Donovan 1991). The line separating these populations has been debated since their acceptance by the IWC's Scientific Committee. For stock assessment purposes, NMFS recognizes three discrete population centers of sperm whales in the Pacific: (1) Alaska, (2) California-Oregon-Washington, and (3) Hawaii.

Sperm whales are widely distributed throughout the Hawaiian Islands throughout the year and are the most abundant large whale in waters off Hawai'i during the summer and fall (Rice 1960, Shallenberger 1981, Lee 1993, and Mobley *et al.* 2000). Sperm whale clicks recorded from hydrophones off Oahu confirm the presence of sperm whales near the Hawaiian Islands throughout the year (Thompson and Friedl 1982). The primary area of occurrence for the sperm whale is seaward of the shelf break in the Hawaiian Islands.

Sperm whales south of the equator are generally treated as a single "population," although the International Whaling Commission divides these whales into nine different divisions that are based more on evaluations of whaling captures than the biology of sperm whales (Donovan 1991). Several authors, however, have argued that the sperm whales that occur off the Galapagos Islands, mainland Ecuador, and northern Peru are geographically distinct from other sperm whales in the Southern Hemisphere (Rice 1977, Wade and Gerrodette 1993, and Dufault and Whitehead 1995).

Threats to the Species

Natural threats. Sperm whales are hunted by killer whales (*Orcinus orca*), false killer whales (*Pseudorca crassidens*), and short-finned pilot whales (*Globicephala melas*; Arnbom *et al.* 1987, Palacios and Mate 1996, Rice 1989, Weller *et al.* 1996, Whitehead 1995). Sperm whales have been observed with bleeding wounds on their heads and tail flukes after attacks by these species (Arnbom *et al.* 1987, Dufault and Whitehead 1995).

Studies on sperm whales in the North Pacific and North Atlantic Oceans have demonstrated that sperm whales are infected by caliciviruses and papillomavirus (Smith and Latham 1978, Lambertsen *et al.* 1987). In some instances, these diseases have been demonstrated to affect 10 percent of the sperm whales sampled (Lambertsen *et al.* 1987).

Anthropogenic threats. Three human activities are known to threaten sperm whales: whaling, entanglement in fishing gear, and shipping. Historically, whaling represented the greatest threat to every population of sperm whales and was ultimately responsible for listing sperm whales as an endangered species. Sperm whales were hunted all over the world during the 1800s, largely for spermaceti oil and ambergris. Harvesting of sperm whales subsided by 1880 when petroleum replaced the need for sperm whale oil (Whitehead 2003).

The actual number of sperm whales killed by whalers remains unknown and some of the estimates of harvest numbers are contradictory. Between 1800 and 1900, the International Whaling Commission estimated that nearly 250,000 sperm whales were killed globally by whalers. From 1910 to 1982, another 700,000 sperm whales were killed globally by whalers (IWC Statistics 1959-1983). These estimates are substantially higher than a more recent estimate produced by Caretta *et al.* (2005), however, who estimated that at least 436,000 sperm whales were killed by whalers between 1800 and 1987. Hill and DeMaster (1999) concluded that about 258,000 sperm whales were harvested in the North Pacific between 1947 and 1987 by commercial whalers. They reported that catches in the North Pacific increased until 1968, when 16,357 sperm whales were harvested, then declined after 1968 because of harvest limits imposed by the IWC. Perry *et al.* (1999) estimated that, on average, more than 20,000 sperm whales were harvested in the Southern Hemisphere each year between 1956 and 1976.

For the Atlantic Four hundred twenty-four sperm whales were harvested in the Newfoundland-Labrador area between 1904 and 1972 and 109 male sperm whales were taken near Nova Scotia in 1964-1972 (Mitchell and Kozicki 1984) in a Canadian whaling fishery. There was also a well-documented sperm whale fishery based on the west coast of Iceland. Other sperm whale catches occurred near West

Greenland, the Azores, Madeira, Spain, Spanish Morocco, Norway (coastal and pelagic), the Faroes, and Britain. At present, because of their general offshore distribution, sperm whales are less likely to be impacted by humans and those impacts that do occur are less likely to be recorded. There has been no complete analysis and reporting of existing data on this topic for the western North Atlantic.

These reports probably underestimate the actual number of sperm whales that were killed by whalers, particularly because they could not have incorporated realistic estimates of the number of sperm whales killed by Soviet whaling fleets, which often went unreported. Between 1947 and 1973, Soviet whaling fleets engaged in illegal whaling in the Indian, North Pacific, and southern Oceans. In the Southern Hemisphere, these whalers killed an estimated 100,000 whales that they did not report to the International Whaling Commission (Yablokov *et al.* 1998). Illegal catches in the Northern Hemisphere (primarily in the North Pacific) were smaller but still caused sperm whales to disappear from large areas of the North Pacific Ocean (Yablokov and Zemsky 2000).

In addition to large and illegal harvests of sperm whales, Soviet whalers had disproportionate effect on sperm whale populations because they commonly killed adult females in any reproductive condition (pregnant or lactating) as well as immature sperm whales of either gender.

When the International Whaling Commission introduced the International Observer Scheme in 1972, the IWC relaxed regulations that limited the minimum length of sperm whales that could be caught from 11.6 meters to 9.2 meters out of a concern that too many male sperm whales were being caught so reducing this size limit would encourage fleets to catch more females. Unfortunately, the IWC's decision had been based on data from the Soviet fleets who commonly reported female sperm whales as males. As a result, the new regulations allowed the Soviet whalers to continue their harvests of female and immature sperm whales legally, with substantial consequences for sperm whale populations. Berzin noted in a report he wrote in 1977, "the result of this was that some breeding areas for sperm whales became deserts" (Berzin 2007).

In U.S. waters in the Pacific Ocean, sperm whales are known to have been incidentally captured only in drift gillnet operations, which killed or seriously injured an average of 9 sperm whales per year from 1991 - 1995 (Barlow *et al.* 1997). Interactions between longline fisheries and sperm whales in the Gulf of Alaska have been reported over the past decade (Rice 1989, Hill and DeMaster 1999). Observers aboard Alaskan sablefish and halibut longline vessels have documented sperm whales feeding on fish caught in longline gear in the Gulf of Alaska. During 1997, the first entanglement of a sperm whale in Alaska's longline fishery was recorded, although the animal was not seriously injured (Hill and DeMaster 1998).

The available evidence does not indicate sperm whales are being killed or seriously injured as a result of these interactions, although the nature and extent of interactions between sperm whales and long-line gear is not yet clear.

Several sperm whale entanglements have been documented in the North Atlantic. In July 1990, a sperm whale was entangled and subsequently released (injured) from the now prohibited pelagic drift gillnet near the continental shelf edge on southern Georges Bank. In August 1993, a dead sperm whale, with longline gear wound tightly around the jaw, was found floating about 20 miles off Mt Desert Rock. In October 1994, a sperm whale was successfully disentangled from a fine- mesh gillnet in Birch Harbor, Maine. During June 1995, one sperm whale was entangled with "gear in/around several body parts" then released injured from a pelagic drift gillnet haul located on the shelf edge between Oceanographer and Hydrographer Canyons on Georges Bank. In May 1997, a sperm whale entangled in net with three buoys trailing was sighted 130 nm northwest of Bermuda. No information on the status of the animal was provided.

Sperm whales are also killed by ship strikes. In May 1994 a sperm whale that had been struck by a ship was observed south of Nova Scotia (Reeves and Whitehead 1997) and in May 2000 a merchant ship reported a strike in Block Canyon (NMFS, unpublished data), which is a major pathway for sperm whales entering southern New England continental shelf waters in pursuit of migrating squid (CeTAP 1982, Scott and Sadove 1997). One sperm whale died as the result of a ship strike in Oregon in 2007 (NMFS Northwest Regional Stranding data, unpublished).

Status

Sperm whales were listed as endangered under the ESA in 1973. Sperm whales have been protected from commercial harvest by the International Whaling Commission since 1981, although the Japanese continued to harvest sperm whales in the North Pacific until 1988 (Reeves and Whitehead 1997). They are also protected by the Convention on International Trade in Endangered Species of Wild Flora and Fauna and the MMPA. Critical habitat has not been designated for sperm whales.

The status and trend of sperm whales at the time of this summary is largely unknown. Allen and Angliss (2012) reported that estimates for population abundance, status, and trends for sperm whales off the coast of Alaska were not available when they prepared the Stock Assessment Report for marine mammals off Alaska. Similarly, no information was available to support estimates of sperm whales status and trends in the western North Atlantic Ocean (Waring *et al.* 2007), the Indian Ocean (Perry *et al.* 1999), or the Mediterranean Sea.

Nevertheless, several authors and organizations have published “best estimates” of the global abundance of sperm whales or their abundance in different geographic areas. Based on historic whaling data,190,000 sperm whales were estimated to have been in the entire North Atlantic, but the IWC considers data that produced this estimate unreliable (Perry *et al.* 1999). Whitehead (2002) estimated that prior to whaling sperm whales numbered around 1,110,000 and that the current global abundance of sperm whales is around 360,000 (coefficient of variation = 0.36) whales. Whitehead’s current population estimate (2002) is about 20% of past global abundance estimates which were based on historic whaling data.

Waring *et al.* (2007) concluded that the best estimate of the number of sperm whales along the Atlantic coast of the U.S. was 4,804 (coefficient of variation = 0.38) in 2004, with a minimum estimate of 3,539 sperm whales in the western North Atlantic Ocean.

The information available on the status and trend of sperm whales do not allow us to make a definitive statement about the extinction risks facing sperm whales as a species or particular populations of sperm whales. However, the evidence available suggests that sperm whale populations probably exhibit the dynamics of small populations, causing their population dynamics to become a threat in and of itself. The number of sperm whales killed by Soviet whaling fleets in the 1960s and 1970s would have substantial and adverse consequence for sperm whale populations and their ability to recover from the effects of whaling on their population. The number of adult females killed by Soviet whaling fleets, including pregnant and lactating females whose death would also have resulted in the death of their calves, would have had a devastating effect on sperm whale populations. In addition to decimating their population size, whaling would have skewed sex ratios in their populations, created gaps in the age structure of their populations, and would have had lasting and adverse effect on the ability of these populations to recover (for example, see Whitehead 2003).

Data regarding the abundance and trends for sperm whales in the Atlantic are sparse. An abundance of 2,607 (CV=0.57) for sperm whales was estimated from a line-transect sighting survey conducted 345 during 12 June to 4 August 2004 by a ship and plane that surveyed 10,761 km of track line in waters north of Maryland (about 38ºN) to the Bay of Fundy (about 45ºN) (Palka 2006).

A survey of the U.S. Atlantic outer continental shelf and continental slope (water depths>50 m) between Florida and Maryland (27.5 and 38ºN) was conducted during June-August, 2004. Survey effort was stratified to include increased effort along the continental shelf break and Gulf Stream front in the mid-Atlantic. The survey included 5,659 km of trackline, and there were a total of 473 cetacean sightings. Sightings were most frequent in waters north of Cape Hatteras, North Carolina along the shelf break. The resulting abundance estimate for sperm whales between Florida and Maryland was 2,197 (CV=0.47*)* (Waring et al 2010).

Populations of sperm whales could not have recovered from the overharvests of adult females and immature whales in the 30 to 40 years that have passed since the end of whaling, but the information available does not allow us to determine whether and to what degree those populations might have stabilized or whether they have begun the process of recovering from the effects of whaling. Absent information to the contrary, we assume that sperm whales will have elevated extinction probabilities because of both exogenous threats caused by anthropogenic activities (primarily whaling, entanglement, and ship strikes) and natural phenomena (such as disease, predation, or changes in the distribution and abundance of their prey in response to changing climate) as well as endogenous threats caused by the legacy of overharvests of adult females and immature whales on their populations (that is, a population with a disproportion of adult males and older animals coupled with a small percentage of juvenile whales that recruit into the adult population).

Diving and Social Behavior

Sperm whales are probably the deepest and longest diving mammal: they can dive to depths of at least 2000 meters (6562 ft), and may remain submerged for an hour or more (Watkins *et al.* 1993). Typical foraging dives last 40 min and descend to about 400 m followed by about 8 min of resting at the surface (Gordon 1987; Papastavrou *et al.* 1989). However, dives of over 2 hr and as deep as 3,000 m have been recorded (Clarke 1976; Watkins *et al.* 1985). Descent rates recorded from echo-sounders were approximately 1.7m/sec and nearly vertical (Goold and Jones 1995). There are no data on diurnal differences in dive depths in sperm whales. However, like most diving vertebrates for which there are data (e.g. rorqual whales, fur seals, chinstrap penguins), sperm whales probably make relatively shallow dives at night when organisms from the ocean’s deep scattering layers move toward the ocean’s surface.

The groups of closely related females and their offspring develop dialects specific to the group (Weilgart and Whitehead 1997) and females other than birth mothers will guard young at the surface (Whitehead 1996) and will nurse young calves (Reeves and Whitehead 1997).

Vocalizations and Hearing

Sperm whales produce loud broad-band clicks from about 0.1 to 20 kHz (Weilgart and Whitehead 1993, 1997; Goold and Jones 1995). These have source levels estimated at 171 dB re 1 μPa (Levenson 1974). Current evidence suggests that the disproportionately large head of the sperm whale is an adaptation to produce these vocalizations (Norris and Harvey 1972; Cranford 1992; but see Clarke 1979). This suggests that the production of these loud low frequency clicks is extremely important to the survival of individual sperm whales. The function of these vocalizations is relatively well-studied (Weilgart and Whitehead 1993, 1997; Goold and Jones 1995). Long series of monotonous regularly spaced clicks are associated with feeding and are thought to be produced for echolocation. Distinctive, short, patterned series of clicks, called codas, are associated with social behavior and intragroup interactions; they are thought to facilitate intra-specific communication, perhaps to maintain social cohesion with the group (Weilgart and Whitehead 1993).

A general description of the anatomy of the ear for cetaceans is provided in the description of the blue whale above. The only data on the hearing range of sperm whales are evoked potentials from a stranded neonate (Carder and Ridgway 1990). These data suggest that neonatal sperm whales respond to sounds from 2.5-60 kHz. Sperm whales have been observed to frequently stop echolocating in the presence of underwater pulses made by echosounders and submarine sonar (Watkins and Schevill 1975; Watkins *et al.* 1985). 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). Sperm whales have moved out of areas after the start of air gun seismic testing (Davis *et al.* 1995). Seismic air guns produce loud, broadband, impulsive noise (source levels can be on the order of 250 dB) with “shots” fired typically every 15 seconds, 240 shots per hour, 24 hours per day during active tests. 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). Furthermore, because of their apparent role as important predators of mesopelagic squid and fish, changing the abundance of sperm whales should affect the distribution and abundance of other marine species.

Green Sea Turtle

Distribution

Green sea turtles are distributed circumglobally, and can be found in the Pacific, Indian, and Atlantic Oceans, Caribbean Sea, as well as the Mediterranean Sea (NMFS and USFWS 1991, 2007d; Seminoff 2004), primarily in tropical or, to a lesser extent, subtropical waters. These regions can be further divided into nesting aggregations within the eastern, central, and western Pacific Ocean; the western, northern, and eastern Indian Ocean; Mediterranean Sea; and eastern, southern, and western Atlantic Ocean, including the Caribbean Sea.

Green turtles appear to prefer waters that usually remain around 20°C in the coldest month. During warm spells (e.g., El Niño), green turtles may be found considerably north of their normal distribution. Stinson (1984) found green turtles appear most frequently in U.S. coastal waters with temperatures exceeding 18°C. An east Pacific green turtle equipped with a satellite transmitter was tracked along the California coast and showed a distinct preference for waters with temperatures above 20°C (Eckert, unpublished data).

Further, green sea turtles seem to occur preferentially in drift lines or surface current convergences, probably because of the prevalence of cover and higher densities of their food items associated with these oceanic phenomena. For example, in the western Atlantic Ocean, drift lines commonly contain floating *Sargassum* capable of providing small turtles with shelter and sufficient buoyancy to raft upon (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 from natural predators and humans. Available information indicates that green turtle resting areas are in proximity to their feeding pastures (NMFS 2000).

Pacific Ocean

Green sea turtles occur in the western, central, and eastern Pacific. Foraging areas are located throughout the Pacific and along the southwestern U.S. coast (NMFS and USFWS 1998b). In the western Pacific, major nesting rookeries at four sites including Heron Island (Australia), Raine Island (Australia), Guam, and Japan were evaluated. Three where determined to be increasing in abundance, while the population in Guam appears stable (NMFS and USFWS 2007d). In the central Pacific, nesting occurs on French Frigate Shoals, Hawaii, which has also been reported as increasing (NMFS and USFWS 2007d). The main nesting sites for green sea turtles in the eastern Pacific are located in Michoacan, Mexico and in the Galapagos Islands, Ecuador (NMFS and USFWS 2007d). The number of nesting females per year exceeds 1,000 females at each site (NMFS and USFWS 2007d). However, historically, more than 20,000 females per year are believed to have nested in Michoacan alone (Cliffton et al. 1982; NMFS and USFWS 2007d). The Pacific Mexico green turtle nesting population (also called the black turtle) is considered endangered.

Historically, green sea turtles were caught for food in many areas of the Pacific. They also were commercially exploited, which, coupled with habitat degradation, led to their decline in the Pacific (NMFS and USFWS 1998b). Green sea turtles in the Pacific continue to be affected by poaching, habitat loss or degradation, fishing gear interactions, and fibropapillomatosis, which is a viral disease that causes tumors in affected turtles (NMFS and USFWS 1998b; NMFS 2004d).

Indian Ocean

There are numerous nesting sites for green sea turtles in the Indian Ocean. One of the largest nesting sites for green sea turtles worldwide occurs on the beaches of Oman where an estimated 20,000 green sea turtles nest annually (Hirth 1997; Ferreira et al. 2003). Based on a review of the 32 Index Sites used to monitor green sea turtle nesting worldwide, Seminoff (2004) concluded that declines in green sea turtle nesting were evident for many of the Indian Ocean Index Sites. While several of these had not demonstrated further declines in the recent past, only the Comoros Island Index Site in the western Indian Ocean showed evidence of increased nesting (Seminoff 2004).

Mediterranean Sea

There are four nesting concentrations of green sea turtles in the Mediterranean from which data are available – Turkey, Cyprus, Israel, and Syria. Currently, approximately 300-400 females nest each year, about two-thirds of which nest in Turkey and one-third in Cyprus. Although green sea turtles are depleted from historic levels in the Mediterranean Sea (Kasparek et al. 2001), nesting data gathered since the early 1990s in Turkey, Cyprus, and Israel show no apparent trend in any direction. However, a declining trend is apparent along the coast of Palestine/Israel, where 300-350 nests were deposited each year in the 1950s (Sella 1982) compared to a mean of six nests per year from 1993-2004 (Kuller 1999; Y. Levy, Israeli Sea Turtle Rescue Center, unpublished data). A recent discovery of green sea turtle nesting in Syria adds roughly 100 nests per year to green sea turtle nesting activity in the Mediterranean (Rees et al. 2005). That such a major nesting concentration could have gone unnoticed until recently (the Syrian coast was surveyed in 1991, but nesting activity was attributed to loggerheads) bodes well for the speculation that the unsurveyed coast of Libya may also host substantial nesting.

Atlantic Ocean

In the western Atlantic, large juvenile and adult green sea turtles are largely herbivorous, occurring in habitats containing benthic algae and seagrasses from Massachusetts to Argentina, including the Gulf of Mexico and Caribbean (Wynne and Schwartz 1999). Green sea turtles occur seasonally in Mid-Atlantic and Northeast waters such as Chesapeake Bay and Long Island Sound (Musick and Limpus 1997; Morreale and Standora 1998; Morreale et al. 2005), which serve as foraging and developmental habitats.

Some of the principal feeding areas in the western Atlantic Ocean include the upper west coast of Florida, the Florida Keys, and the northwestern coast of the Yucatán Peninsula. Additional important foraging areas in the western Atlantic include the Mosquito and Indian River Lagoon systems and nearshore wormrock reefs between Sebastian and Ft. Pierce Inlets in Florida, Florida Bay, the Culebra archipelago and other Puerto Rico coastal waters, the south coast of Cuba, the Mosquito Coast of Nicaragua, the Caribbean coast of Panama, and scattered areas along Colombia and Brazil (Hirth 1971).

Population Structure

The population dynamics of green sea turtles and all of the other sea turtles we consider here are usually described based on the distribution and habit of nesting females, rather than their male counterparts. The spatial structure of male sea turtles and their fidelity to specific coastal areas is unknown; however, we describe sea turtle populations based on the nesting beaches that female sea turtles return to when they mature. Because the patterns of increase or decrease in the abundance of sea turtle nests over time are determined by internal dynamics rather than external dynamics, we make inferences about the growth or decline of sea turtle populations based on the status and trend of their nests.

Primary nesting aggregations of green turtles (i.e. sites with greater than 500 nesting females per year) include: Ascension Island (south Atlantic Ocean), Australia, Brazil, Comoros Islands, Costa Rica, Ecuador (Galapagos Archipelago), Equatorial Guinea (Bioko Island), Guinea-Bissau (Bijagos Archipelago), Iles Eparses Islands (Tromelin Island, Europa Island), Indonesia, Malaysia, Myanmar, Oman, Philippines, Saudi Arabia, Seychelles Islands, Suriname, and United States (Florida; Seminoff 2002, NMFS and USFWS 1998a).

Smaller nesting aggregations include: Angola, Bangladesh, Bikar Atoll, Brazil, Chagos Archipelago, China, Costa Rica, Cuba, Cyprus, Democratic Republic of Yemen, Dominican Republic, d'Entrecasteaux Reef, French Guiana, Ghana, Guyana, India, Iran, Japan, Kenya, Madagascar, Maldives Islands, Mayotte Archipelago, Mexico, Micronesia, Pakistan, Palmerston Atoll, Papua New Guinea, Primieras Islands, Sao Tome é Principe, Sierra Leone, Solomon Islands, Somalia, Sri Lanka, Taiwan, Tanzania, Thailand, Turkey, Scilly Atoll, United States (Hawai'i), Venezuela, and Vietnam (Seminoff 2002).

Molecular genetics techniques have helped researchers gain insight into the distribution and ecology of migrating and nesting green turtles. In the Pacific Ocean, green sea turtles group into two distinct regional clades: (1) western Pacific and South Pacific islands, and (2) eastern Pacific and central Pacific, including the rookery at French Frigate Shoals, Hawai'i. In the eastern Pacific, greens forage coastally from San Diego Bay, California in the north to Mejillones, Chile in the South. Based on mtDNA analyses, green turtles found on foraging grounds along Chile's coast originate from the Galapagos nesting beaches, while those greens foraging in the Gulf of California originate primarily from the Michoacan nesting stock. Green turtles foraging in San Diego Bay and along the Pacific coast of Baja California originate primarily from rookeries of the Islas Revillagigedos (Dutton 2003).

Threats to the Species

Natural Threats. Green sea turtles face many of the same natural threats as loggerhead and Kemp's ridley sea turtles. In addition, green sea turtles appear to be particularly susceptible to fibropapillomatosis, an epizootic disease producing lobe-shaped tumors on the soft portion of a turtle's body and often fatal. Juveniles appear to have the highest incidence of disease and the most extensive lesions, whereas lesions in nesting adults are rare. Also, green sea turtles frequenting nearshore waters, areas adjacent to large human populations, and areas with low water turnover, such as lagoons, have a higher incidence of the disease than individuals in deeper, more remote waters. The occurrence of fibropapilloma tumors may result in impaired foraging, breathing, or swimming ability, leading potentially to death (George 1997).

Green turtles in the northwest Hawai'ian Islands are afflicted with fibropapilloma, as well as spirochidiasis, both of which are the major causes of strandings of this species. The presence of fibropapillomatosis among stranded turtles has increased significantly over the past 17 years, ranging from 47-69 percent during the past decade (Murakawa *et al.* 2000). Green turtles captured off Molokai from 1982-96 showed a massive increase in the disease over this period, peaking at 61% prevalence in 1995 (Balazs *et al.* 1998). Preliminary evidence suggests an association between the distribution of fibropapillomatosis in the Hawai'ian Islands and the distribution of toxic benthic dinoflagellates (*Prorocentrum* spp.) known to produce a tumor promoter, okadaic acid (Landsberg *et al.* 1999). Fibropapillomatosis is considered to decrease growth rates in afflicted turtles and may inhibit the growth rate of Hawai'ian green turtle populations (Balazs *et al.* 1998).

Anthropogenic Threats. Three human activities are known to threaten green sea turtles: overharvests of individual animals, incidental capture in commercial fisheries, and human development of coastlines. Historically, the primary cause of the global decline of green sea turtles populations were the number of eggs and adults captured and killed on nesting beaches in combination with the number of juveniles and adults captured and killed in coastal feeding areas. Some populations of green sea turtles still lose large numbers of eggs, juveniles, and adults to subsistence hunters, local communities that have a tradition of harvesting sea turtles, and poachers in search of turtle eggs and meat.

Green sea turtles were once the target of directed fisheries in the United States and throughout the Caribbean. In 1890, over one million pounds of green sea turtles were taken in a directed fishery in the Gulf of Mexico (Doughty 1984). However, declines in the turtle fishery throughout the Gulf of Mexico were evident by 1902 (Doughty 1984).

Directed harvests of eggs and other life stages of green sea turtles were identified as a “major problem” in American Samoa, Guam, Palau, Commonwealth of the Northern Mariana Islands, Federated States of Micronesia, Republic of the Marshall Islands, and the Unincorporated Islands (Wake, Johnston, Kingman, Palmyra, Jarvis, Howland, Baker, and Midway). In the Atlantic, green sea turtles are captured and killed in turtle fisheries in Colombia, Grenada, the Lesser Antilles, Nicaragua, St. Vincent and the Grenadines (Bräutigam and Eckert 2006); the turtle fishery along the Caribbean coast of Nicaragua has captured more than 11,000 green sea turtles each year for the past 10 years (Bräutigam and Eckert 2006, Lagueux 1998).

Severe overharvests have resulted from a number of factors in modern times: (1) the loss of traditional restrictions limiting the number of turtles taken by island residents; (2) modernized hunting gear; (3) easier boat access to remote islands; (4) extensive commercial exploitation for turtle products in both domestic markets and international trade; (5) loss of the spiritual significance of turtles; (6) inadequate regulations; and (7) lack of enforcement (NMFS and USFWS 1998a).

As with the other sea turtle species, incidental fishery mortality accounts for a large proportion of annual human-caused mortality outside the nesting beaches. Witherington et al. (2009) observes that because green sea turtles spend a shorter time in oceanic waters and as older juveniles occur on shallow seagrass pastures (where benthic trawling is unlikely), they avoid high mortalities in pelagic longline and benthic trawl fisheries. Although the relatively low number of observed green sea turtle captures makes it difficult to estimate bycatch rates and annual take levels, green sea turtles have been observed captured in the pelagic driftnet, pelagic longline, traps and pots, southeast shrimp trawl, and mid-Atlantic trawl and gillnet fisheries. Gillnets account for the highest number of green sea turtles that are captured and killed. Murray (2009a) also lists five observed captures of green turtle in Mid-Atlantic sink gillnet gear between 1995 and 2006.

Finkbeiner et al. (2011) compiled cumulative sea turtle bycatch information in U.S. fisheries from 1990 through 2007, before and after implementation of bycatch mitigation measures. Information was obtained from peer reviewed publications and NMFS documents (e.g., Opinions and bycatch reports). In the Atlantic, a mean estimate of 137,700 bycatch interactions, of which 4,500 were mortalities, occurred annually (since implementation of bycatch mitigation measures). The Southeast/Gulf of Mexico shrimp trawl fishery was responsible for the vast majority of U.S. interactions (up to 98%) and mortalities (more than 80%). While this provides an initial cumulative bycatch assessment, there are a number of caveats that should be considered when interpreting this information, such as sampling inconsistencies and limitations. The most recent section 7 consultation on the shrimp fishery, completed in May 2012, was unable to estimate the total annual level of take for green sea turtles at present. Instead, it qualitatively estimated that the shrimp fishery, as currently operating, would result in at least hundreds and possibly low thousands of interactions annually, of which hundreds are expected to be lethal (NMFS 2012b).

Other activities like channel dredging, marine debris, pollution, vessel strikes, power plant impingement, and habitat destruction account for an unquantifiable level of other mortality. Stranding reports indicate that between 200-400 green sea turtles strand annually along the eastern U.S. coast from a variety of causes most of which are unknown (STSSN database). The most recent five-year status review for green sea turtles notes that global climate change is affecting the species and will likely continue as a threat to these turtles (NMFS and USFWS 2007d). Climate change may impact nesting beaches through sea level rise which may reduce the availability of nesting habitat and increase the risk of nest inundation as well as skewing sex ratios of sea turtle populations. Loss of appropriate nesting habitat may also be accelerated by a combination of other environmental and oceanographic changes, such as an increase in the frequency of storms and/or changes in prevailing currents, both of which could lead to increased beach loss via erosion. Oceanic changes related to rising water temperatures could result in changes in the abundance and distribution of the primary food sources of green sea turtles, which in turn could result in changes in behavior and distribution of this species. Seagrass habitats may suffer from decreased productivity and/or increased stress due to sea level rise, as well as salinity and temperature changes (Short and Neckles 1999; Duarte 2002).

Status

Green turtles are listed as threatened under the ESA, except for breeding populations found in Florida and the Pacific coast of Mexico, which are listed as endangered. Seminoff (2002) estimates using a conservative approach that the global green turtle population has declined by 34% to 58% over the last three generations (approximately 150 years). Actual declines may be closer to 70% to 80%. Causes for this decline include harvest of eggs, subadults and adults, incidental capture by fisheries, loss of habitat, and disease. Critical habitat was designated in 1998 for green turtles in coastal waters around Culebra Island, Puerto Rico, and its outlying keys.

While some nesting populations of green turtles appear to be stable or increasing in the Atlantic Ocean (e.g. Bujigos Archipelago (Guinea-Bissau), Ascension Island, Tortuguero (Costa Rica), Yucatan Peninsula (Mexico), and Florida), declines of over 50% have been documented in the eastern (Bioko Island, Equatorial Guinea) and western Atlantic (Aves Island, Venezuela). Nesting populations in Turkey (Mediterranean Sea) have declined between 42% and 88% since the late 1970s. Population trend variations also appear in the Indian Ocean. Declines greater than 50% have been documented at Sharma (Republic of Yemen) and Assumption and Aldabra (Seychelles), while no changes have occurred at Karan Island (Saudi Arabia) or at Ras al Hadd (Oman). The number of females nesting annually in the Indian Ocean has increased at the Comoros Islands, Tromelin and maybe Europa Island (Iles Esparses; Seminoff 2002).

Green turtles are thought to be declining throughout the Pacific Ocean, with the exception of Hawaii, as a direct consequence of a historical combination of overexploitation and habitat loss (Eckert 1993, Seminoff 2002). They are also thought to be declining in the Atlantic Ocean. However, like several of the species we have already discussed, the information available on the status and trend of green sea turtles do not allow us to make a definitive statement about the global extinction risks facing these sea turtles or risks facing particular populations (nesting aggregations) of these turtles. With the limited data available on green sea turtles, we do not know whether green sea turtles exist at population sizes large enough to avoid demographic phenomena that are known to increase the extinction probability of species that exist as “small” populations (that is, “small” populations experience phenomena such as demographic stochasticity, inbreeding depression and Allee effects, among others, that cause their population size to become a threat in and of itself) or if green sea turtles are threatened more by exogenous threats such as anthropogenic activities (entanglement, habitat loss, overharvests, etc.) or natural phenomena (such as disease, predation, or changes in the distribution and abundance of their prey in response to changing climate). Nevertheless, with the exception of the Hawaiian nesting aggregations, we assume that green sea turtles are endangered because of both anthropogenic and natural threats as well as changes in their population dynamics.

Diving Behavior

Based on the behavior of post-hatchlings and juvenile green turtles raised in captivity, we presume that those in pelagic habitats live and feed at or near the ocean surface, and that their dives do not normally exceed several meters in depth (Hazel et al. 2009; NMFS and USFWS 1998a). Recent data from Australia indicate green sea turtles rarely dive deep, staying in upper 8 m of the water column (Hazel et al. 2009). Here, daytime dives were shorter and shallower than were nighttime dives. Also, time spent resting and dive duration increased significantly with decreases in seasonal water temperatures. The maximum recorded dive depth for an adult green turtle was just over 106 m (Berkson 1967 *in* Lutcavage and Lutz 1997), while subadults routinely dive to 20 m for 9-23 min, with a maximum recorded dive of over 1 h (Brill et al. 1995; I-Jiunn 2009). Green sea turtles along Taiwan may rest during long, shallow dives (I-Jiunn 2009). Dives by females may be shorter in the period leading up to nesting (I-Jiunn 2009).

Hearing

The information on green turtle hearing is limited. Ridgway *et al.* (1969) studied the auditory evoked potentials of three green sea turtles (in air and through mechanical stimulation of the ear) and concluded that their maximum sensitivity occurred from 300 to 400 Hz with rapid declines for tones at lower and higher frequencies. They reported an upper limit for cochlear potentials without injury of 2000 Hz and a practical limit of about 1000 Hz. This is similar to estimates for loggerhead sea turtles, which had most sensitive hearing between 250 and 1000 Hz, with rapid decline above 1000 Hz (Bartol *et al.* 1999).

In a study of the auditory brainstem responses of subadult green sea turtles, Bartol and Ketten (2006) reported responses to frequencies between 100 and 500 Hz; with highest sensitivity between 200 and 400 Hz. They reported that two juvenile green turtles had hearing sensitivities that were slightly broader in range: they responded to sounds at frequencies from 100 to 800 Hz, with highest hearing sensitivities between 600 to 700 Hz. In a more recent study, Piniak et al. (2012) reported an upper limit for auditory evoked potentials of about 1600 Hz in water with maximum sensitivity between 50 and 400 Hz and a sharp decline in sensitivity after 400 Hz for juvenile green turtles.

Hawksbill Sea Turtle

Distribution

The hawksbill has a circumglobal distribution throughout tropical and, to a lesser extent, subtropical waters of the Atlantic, Indian, and Pacific oceans. Satellite tagged turtles have shown significant variation in movement and migration patterns. In the Caribbean, distance traveled between nesting and foraging locations ranges from a few kilometers to a few hundred kilometers (Byles and Swimmer 1994; Hillis-Starr et al. 2000; Horrocks et al. 2001; Lagueux et al. 2003; Miller et al. 1998; Prieto et al. 2001). Hawksbill sea turtles occur in tropical and subtropical seas of the Atlantic, Pacific and Indian Oceans. The species is widely distributed in the Caribbean Sea and western Atlantic Ocean, with individuals from several life history stages occurring regularly along southern Florida and the northern Gulf of Mexico (especially Texas)(Amos 1989); in the Greater and Lesser Antilles; and along the Central American mainland south to Brazil.

In the continental U.S., hawksbill sea turtles have been reported in every state on the coast of the Gulf of Mexico and along the coast of the Atlantic Ocean from Florida to Massachusetts, except for Connecticut (Wallace et al. 2010); however, sightings of hawksbill sea turtles north of Florida are rare. The only states where hawksbill sea turtles occur with any regularity are Florida (particularly in the Florida Keys and the reefs off Palm Beach County on Florida's Atlantic coast, where the warm waters of the Gulf Stream pass close to shore) and Texas (Bowen et al. 2007). In both of these states, most sightings are of post-hatchlings and juveniles that are believed to have originated from nesting beaches in Mexico.

Hawksbill sea turtles have stranded along almost the entire Atlantic coast of the United States, although most stranding records occur south of Cape Canaveral, Florida, particularly in Palm Beach, Broward and Miami-Dade counties (Florida Sea Turtle Stranding and Salvage database). Hawksbill sea turtles are very rare north of Florida, although they have been recorded as far north as Massachusetts. During their pelagic-stage, hawksbills disperse from the Gulf of Mexico and southern Florida in the Gulfstream Current, which could carry them offshore of Georgia and the Carolinas. As evidence of this, a pelagic-stage hawksbill was captured 37 nautical miles east of Sapelo Island, Georgia in May 1994 (Parker 1995). There are also records of hawksbill sea turtles stranding on the coast of Georgia (Ruckdeschel *et al.* 2000), being captured in pound nets off Savannah, and being captured in summer flounder trawls (Epperly *et al.* 1995), gillnets (Epperly *et al.* 1995), and power plants off Georgia and the Carolinas.

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

Hawksbill sea turtles occupy different habitats depending on their life history stage. After entering the sea, hawksbill sea turtles occupy pelagic waters and occupy weedlines that accumulate at convergence points. When they grow to about 20-25 cm carapace length, hawksbill sea turtles reenter coastal waters where they inhabit and forage in coral reefs as juveniles, subadults and adults. Hawksbill sea turtles also occur around rocky outcrops and high energy shoals, where sponges grow and provide forage, and they are known to inhabit mangrove-fringed bays and estuaries, particularly along the eastern shore of continents where coral reefs are absent (Hildebrand 1987, Amos 1989).

Population Structure

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

Threats to the Species

Natural Threats. The various habitat types hawksbill sea turtles occupy throughout their lives exposes these sea turtles to a wide variety of natural threats. The beaches on which hawksbill sea turtles nest and the nests themselves are threatened by hurricanes and tropical storms as well as the storm surges, sand accretion, and rainfall that are associated with hurricanes. Hatchlings are hunted by predators like herons, gulls, dogfish, and sharks. Adult hawksbill sea turtles are also killed by sharks and other large, marine predators.

Anthropogenic Threats. Three human activities are known to threaten hawksbill sea turtles: overharvests of individual animals, incidental capture in commercial fisheries, and human development of coastlines. Historically, the primary cause of the global decline of hawksbill sea turtle populations was overharvest by humans for subsistence and commercial purposes. In the Atlantic, hawksbill sea turtles are still captured and killed in turtle fisheries in Colombia, Grenada, the Lesser Antilles, Nicaragua, St. Vincent and the Grenadines (Bräutigam and Eckert 2006). Climate change may impact nesting beaches through sea level rise which may reduce the availability of nesting habitat and increase the risk of nest inundation as well as skewing sex ratios of sea turtle populations. Loss of appropriate nesting habitat may also be accelerated by a combination of other environmental and oceanographic changes, such as an increase in the frequency of storms and/or changes in prevailing currents, both of which could lead to increased beach loss via erosion. Oceanic changes related to rising water temperatures could result in changes in the abundance and distribution of the primary food sources of hawksbill sea turtles, which in turn could result in changes in behavior and distribution of this species.

For centuries, hawksbill sea turtles have been captured for their shells, which have commercial value, rather than food (the meat of hawksbill sea turtles is considered to have a bad taste and can be toxic to humans; NMFS and USFWS 1998b). Until recently, tens of thousands of hawksbills were captured and killed each year to meet demand for jewelry, ornamentation, and whole stuffed turtles (Milliken and Tokunaga *1987 cited in* Eckert 1993). In 1988, Japan's imports from Jamaica, Haiti and Cuba represented some 13,383 hawksbills: it is extremely unlikely that this volume could have originated solely from local waters (Greenpeace 1989 *in* Eckert 1993).

Although Japan banned the importation of turtle shell in 1994, domestic harvests of eggs and turtles continue in the United States, its territories, and dependencies, particularly in the Caribbean and Pacific Island territories. Large numbers of nesting and foraging hawksbill sea turtles are captured and killed for trade in Micronesia, the Mexican Pacific coast, southeast Asia and Indonesia (NMFS and USFWS 1998b). In addition to the demand for the hawksbill's shell, there is a demand for other products including leather, oil, perfume, and cosmetics. Before the U.S. certified Japan under the Pelly Amendment, Japan had been importing about 20 metric tons of hawksbill shell per year, representing approximately 19,000 turtles.

The second most important threat to hawksbill sea turtles is the loss of nesting habitat caused by the expansion of resident human populations in coastal areas of the world and increased destruction or modification of coastal ecosystems to support tourism. Hawksbill sea turtles are also captured and killed in commercial fisheries. Along the Atlantic coast of the U.S., NMFS estimated that about 650 hawksbill sea turtles are captured in shrimp trawl fisheries each year in the Gulf of Mexico, with most of those sea turtles dying as a result of their capture. Each year, about 35 hawksbill sea turtles are captured in Atlantic pelagic longline fisheries. Although most of these turtles are released alive, these fisheries are expected to kill about 50 hawksbill sea turtles each year; the health effects of being captured on the sea turtles that survive remain unknown.

Like green sea turtles, hawksbill sea turtles are threatened by domestic or domesticated animals that prey on their nests; artificial lighting that disorients adult female and hatchling sea turtles, which can dramatically increase the mortality rates of hatchling sea turtles; beach replenishment; ingestion and entanglement in marine debris; and environmental contaminants.

Status

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. Under Appendix I of the Convention on International Trade in Endangered Species of Wild Fauna and Flora, hawksbill sea turtles are identified as "most endangered" (Baillie and Groombridge, 1996; Meylan and Donnelly, 1999). Critical habitat was designated in 1998 for hawksbill turtles in coastal waters surrounding Mona and Monito Islands, Puerto Rico.

Hawksbill sea turtles are solitary nesters, which makes it difficult to estimate the size of their populations. There are no global estimates of the number of hawksbill sea turtles, but a minimum of 15,000 to 25,000 females are thought to nest annually in more than 60 geopolitical entities (Groombridge and Luxmoore 1989). Moderate populations appear to persist around the Solomon Islands, northern Australia, Palau, Persian Gule islands, Oman, and parts of the Seychelles (Groombridge 1982). In a more recent review, Groombridge and Luxmoore (1989) list Papua New Guinea, Queensland, and Western Australia as likely to host 500-1,000 nesting females per year, while Indonesia and the Seychelles may support >1,000 nesting females. The largest known nesting colony in the world is located on Milman Island, Queensland, Australia where Loop (1995) tagged 365 hawksbills nesting within an 11 week period. With the exception of Mexico, and possibly Cuba, nearly all Wider Caribbean countries are estimated to receive <100 nesting females per year (Meylan 1989).

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 2007b). Worldwide, an estimated 21,212-28,138 hawksbills nest each year among 83 sites. Among the 58 sites with historic trends, all show a decline during the past 20 to 100 years. Among 42 sites for which recent trend data are available, 10 (24 percent) are increasing, three (7 percent) are stable and 29 (69 percent) are decreasing. Encouragingly, nesting range along Mexico and Central America appears not to have contracted and estimates continue to increase as additional dedicated study is conducted in the eastern Pacific (Gaos et al. 2010).

Of the 65 geopolitical units on which hawksbill sea turtles nest and where hawksbill nesting densities can be estimated, 38 geopolitical units have hawksbill populations that are suspected or known to be declining. Another 18 geopolitical units have experienced well-substantiated declines (NMFS and USFWS 1995). The largest remaining nesting concentrations occur on remote oceanic islands off Australia (Torres Strait) and the Indian Ocean (Seychelles) (McCann 2010).

Hawksbill sea turtles, like green sea turtles, are thought to be declining globally as a direct consequence of a historical combination of overexploitation and habitat loss. However, like several of the species we have already discussed, the information available on the status and trend of hawksbill sea turtles does not allow us to make definitive statements about the global extinction risks facing these sea turtles or the risks facing particular populations (nesting aggregations) of these turtles. However, the limited data available suggests that several hawksbill sea turtles populations exist at sizes small enough to be classified as "small" populations (that is, populations that exhibit population dynamics that increase the extinction probabilities of the species or several of its populations) while others are large enough to avoid these problems. Exogenous threats such as overharvests and entanglement in fishing gear only increase their probabilities of becoming extinct in the foreseeable future.

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

Population increase has been greater in the Insular Caribbean than along the Western Caribbean Mainland or the eastern Atlantic (including Sao Tomé and Equatorial Guinea). Nesting populations of Puerto Rico appeared to be in decline until the early 1990s, but have universally increased during the survey periods. Mona Island now hosts 199-332 nesting females annually, and the other sites combined host 51-85 nesting females annually (R.P. van Dam and C.E. Diez, unpublished data in NMFS and USFWS 2007b, C.E. Diez, Chelonia, Inc., in litt. to J. Mortimer 2006). The U.S. Virgin Islands have a long history of tortoiseshell trade (Schmidt 1916). At Buck Island Reef National Monument, protection has been in force since 1988, and during that time, hawksbill nesting has increased by 143 percent to 56 nesting females annually, with apparent spill over to beaches on adjacent St. Croix (Z. Hillis-Starr, National Park Service, in lett. to J. Mortimer 2006). However, St. John populations did not increase, perhaps due to the proximity of the legal turtle harvest in the British Virgin Islands (Z. Hillis-Starr, National Park Service, in litt. to J. Mortimer 2006). Populations have also been identified in Belize and Brazil as genetically unique (Hutchinson and Dutton 2007). An estimated 50 to 200 nests are laid per year in the Guinea-Bissau (Catry et al. 2009).

Diving Behavior

The duration of foraging dives in hawksbill sea turtles commonly depends on the size of the turtle: larger turtles diving deeper and longer. At a study site in the northern Caribbean, foraging dives were made only during the day and dive durations ranged from 19-26 minutes in duration at depths of 8-10 m. At night, resting dives ranged from 35-47 minutes in duration (Van Dam and Diez, 1997).

Hearing

There is no information on hawksbill sea turtle hearing. However, we assume that their hearing sensitivities will be similar to those of green and loggerhead sea turtles with their best hearing sensitivity in the low frequency range: from 50 to 400 Hz with rapid declines for tones at lower and higher frequencies with an upper limit of about 1600 Hz in water (Piniak *et al*. 2012).

Kemp's Ridley Sea Turtle

Distribution

The Kemp's ridley was formerly known only from the Gulf of Mexico and along the Atlantic coast of the U.S. (NMFS et al. 2011, TEWG 2000), as far north as the Grand Banks (Watson et al. 2004) and Nova Scotia (Bleakney 1955). However, recent records support Kemp's ridley sea turtles distribution extending into the Azores and eastern north Atlantic (Deraniyagala 1938, Brongersma 1972, Fontaine et al. 1989a, Bolten and Martins 1990) and Mediterranean (Pritchard and Marquez 1973, Brongersma and Carr 1983, Tomas and Raga 2008, Insacco and Spadola 2010). The vast majority of individuals stem from breeding beaches at Rancho Nuevo on the Gulf of Mexico coast of Mexico. Females rarely leave the Gulf of Mexico and adult males do not migrate. Juveniles feed along the east coast of the United States up to the waters off Cape Cod, Massachusetts (Spotila 2004). A small number of individuals reach European waters (Brongersma 1972, Spotila 2004) and the Mediterranean (Pritchard and Marquez-M. 1973, Insacco and Spadola 2010).

Juvenile Kemp's ridley sea turtles are the second most abundant sea turtle in the mid-Atlantic region from New England, New York, and the Chesapeake Bay, south to coastal areas off North Carolina (Morreale et al. 2007; TEWG 2000; Schmid 1998; Wibbels et al. 2005). In the fall, they migrate south along the coast, forming one of the densest concentrations of Kemp's ridley sea turtles outside of the Gulf of Mexico (Musick and Limpus 1997).

Developmental habitats are defined by several characteristics, including sheltered coastal areas such as embayments and estuaries, and nearshore temperate waters shallower than 50 meters (NMFS and USFWS 2007c), although it is not uncommon for adults to venture into deeper waters (Byles 1989a; Mysing and Vanselous 1989; Renaud et al. 1996; Shaver et al. 2005; Shaver and Wibbels 2007b). The suitability of these habitats depends on resource availability, with optimal environments providing rich sources of crabs and other invertebrates.

Foraging areas documented along the U.S. Atlantic coast include Charleston Harbor, Pamlico Sound (Epperly et al. 1995c), Chesapeake Bay (Musick and Limpus 1997), Delaware Bay (Stetzar 2002), and Long Island Sound (Morreale and Standora 1993; Morreale et al. 2005). For instance, in the Chesapeake Bay, Kemp's ridleys frequently forage in submerged aquatic grass beds for crabs (Musick and Limpus 1997). Upon leaving Chesapeake Bay in autumn, juvenile Kemp's ridleys migrate down the coast, passing Cape Hatteras in December and January (Musick and Limpus 1997). These larger juveniles are joined by juveniles of the same size from North Carolina and smaller juveniles from New York and New

England to form one of the densest concentrations of Kemp's ridleys outside of the Gulf of Mexico (Epperly et al. 1995a, 1995b; Musick and Limpus 1997).

Population Structure

The majority of Kemp's ridleys nest along a single stretch of beach near Rancho Nuevo, Tamaulipas, Mexico (Carr 1963; NMFS and USFWS 2007c; NMFS et al. 2011). There is a limited amount of scattered nesting to the north and south of the primary nesting beach (NMFS and USFWS 2007c). Nesting often occurs in synchronized emergences termed arribadas. The number of recorded nests reached an estimated low of 702 nests in 1985, corresponding to fewer than 300 adult females nesting in that season (TEWG 2000; NMFS and USFWS 2007c; NMFS et al. 2011). Conservation efforts by Mexican and U.S. agencies have aided this species by eliminating egg harvest, protecting eggs and hatchlings, and reducing at-sea mortality through fishing regulations (TEWG 2000). Since the mid-1980s, the number of nests observed at Rancho Nuevo and nearby beaches has increased 14%-16% per year (Heppell et al. 2005), allowing cautious optimism that the population is on its way to recovery. An estimated 5,500 females nested in the State of Tamaulipas over a three-day period in May 2007 and more than 4,000 of those nested at Rancho Nuevo (NMFS and USFWS 2007c). In 2008, 17,882 nests were documented on Mexican nesting beaches (NMFS 2011b). There is limited nesting in the U.S., most of which is located in south Texas. Occasional nesting has been documented in North Carolina, South Carolina, and the Gulf and Atlantic coasts of Florida.

Threats to the Species

Natural Threats. Kemp's ridley sea turtles are exposed to a wide variety of threats during every stage of their lives. Eggs and hatchlings on nesting beaches are preyed upon by coyotes, raccoons, coatis, skunks, ghost crabs, ants, and to lesser degrees hawks, vultures, grackles, and caracaras (Dodd 1988, Hirth 1971, Witzell 1983). Those hatchlings that reach the ocean are preyed upon by gulls, terns, sharks, and predatory fish (Dodd 1988). Sharks and other large marine predators prey on large juvenile Kemp's ridley sea turtles. Kemp's ridleys also face oceanographic-related events such as coldstunning.

Although cold-stunning can occur throughout the range of the species, it may be a greater risk for sea turtles that use the more northern habitats of Cape Cod Bay and Long Island Sound (2009). In the last five years (2006-2010), the number of cold-stunned turtles on Cape Cod beaches averaged 115 Kemp's ridleys, seven loggerheads, and seven greens (NMFS unpublished data). The numbers ranged from a low in 2007 of 27 Kemp's ridleys, five loggerheads, and five greens to a high in 2010 of 213 Kemp's ridleys, four loggerheads, and 14 greens. Annual cold stun events vary in magnitude; the extent of episodic major cold stunning events may be associated with numbers of turtles using Northeast U.S. waters in a given year, oceanographic conditions, and/or the occurrence of storm events in the late fall. Although many cold-stunned turtles can survive if they are found early enough, these events are a significant source of natural mortality for Kemp's ridleys.

Because of their restricted geographic distribution, the concentration of most nesting activity at one beach, and the frequency of hurricanes in the Gulf of Mexico, hurricanes represent a substantial threat to Kemp's ridley sea turtles. For example, in 1988 Hurricane Gilbert struck the primary nesting beach, destroyed many of the nests, and altered the structure of the nesting beach.

Anthropogenic Threats. Like other sea turtle species, the severe decline in the Kemp's ridley population appears to have been heavily influenced by a combination of exploitation of eggs, impacts from fishery interactions, loss of foraging habitat, and marine pollution. From the 1940s through the early 1960s, nests from Rancho Nuevo were heavily exploited, but beach protection in 1967 helped to curtail this activity (NMFS et al. 2011).

Kemp's ridley sea turtles have been captured and killed by fishing gear in several Federal and state fisheries throughout their range. They have been captured in gear used in lobster fisheries and monkfish fisheries off the northeastern United States, pound net fisheries off eastern Long Island, the mid-Atlantic, and Chesapeake Bay; fisheries for squid, mackerel, butterfish, bluefish, summer flounder, Atlantic herring, weakfish, and the sargassum fishery. The most significant fishery-related threat to Kemp's ridley sea turtles has been the number of sea turtles that have been captured and killed in the shrimp trawl fisheries in the Gulf of Mexico.

Although modifications to shrimp trawls have helped to reduce mortality of Kemp's ridleys, a recent assessment found that the Southeast/Gulf of Mexico shrimp trawl fishery remained responsible for the vast majority of U.S. fishery interactions (up to 98%) and mortalities (more than 80%). Finkbeiner et al. (2011) compiled cumulative sea turtle bycatch information in U.S. fisheries from 1990 through 2007, before and after implementation of bycatch mitigation measures. Information was obtained from peer reviewed publications and NMFS documents (e.g., Opinions and bycatch reports). In the Atlantic, a mean estimate of 137,700 bycatch interactions, of which 4,500 were mortalities, occurred annually (since implementation of bycatch mitigation measures). Kemp's ridleys interacted with fisheries most frequently, with the highest level of mean annual mortality (2,700), followed by loggerheads (1,400), greens (300), and leatherbacks (40). While this provides an initial cumulative bycatch assessment, there are a number of caveats that should be considered when interpreting this information, such as sampling inconsistencies and limitations. The most recent section 7 consultation on the shrimp fishery, completed in May 2012, was unable to estimate the total annual level of take for Kemp's ridleys at present. Instead, it qualitatively estimated that the shrimp fishery, as currently operating, would result in at least tens of thousands and possibly hundreds of thousands of interactions annually, of which at least thousands and possibly tens of thousands are expected to be lethal (NMFS 2012a).

Kemp's ridley non-fisheries interactions have also been observed; for example, the Oyster Creek Nuclear Generating Station in Barnegat Bay, New Jersey, recorded a total of 27 Kemp's ridleys (15 of which were found alive) impinged or captured on their intake screens from 1992-2006 (NMFS 2006b), and interactions with coastal dredging remains a concern. NMFS' biological biological opinions has required non-dredging "windows" to protect nesting females as well as relocation trawls that preceed dredging equipment to relocated turtles out of the dredge path.

The recovery plan for Kemp's ridley sea turtles (NMFS et al. 2011) identifies climate change as a threat; however, as with the other species discussed above, no significant climate change related impacts to Kemp's ridley sea turtles have been observed to date. Atmospheric warming could cause habitat alteration which may change food resources such as crabs and other invertebrates. It may increase hurricane activity, leading to an increase in debris in nearshore and offshore waters, which may result in an increase in entanglement, ingestion, or drowning. In addition, increased hurricane activity may cause damage to nesting beaches or inundate nests with sea water. Atmospheric warming may change convergence zones, currents and other oceanographic features that are relevant to Kemp's ridleys, as well as changes in precipitation regimes and levels of nearshore runoff.

Status

Kemp's ridley sea turtles were listed as endangered on December 2, 1970 (35 FR 18320). Internationally, the Kemp's ridley is considered the most endangered sea turtle (NRC 1990b; USFWS 1999). There is no designated critical habitat for the Kemp's ridley sea turtle.

In 1947, 40,000 female Kemp's ridley sea turtles were observed nesting on the beaches at Rancho Nuevo on a single day (Carr 1963, Hildebrand 1963). By the early 1970s, the estimate of mature female Kemp's ridleys had been reduced to 2,500-5,000 individuals. Between the years of 1978 and 1991 only 200 Kemp's ridleys nested annually (TEWG 2000; USFWS and NMFS 1992). Today the Kemp's ridley population appears to be in the early stages of recovery. Nesting has increased steadily over the past decade and the total annual number of nests recorded at Rancho Nuevo and adjacent camps has exceeded 10,000 in recent years. Over 20,000 nests were recorded in 2009 at Rancho Nuevo and adjacent camps.

From 2002-2009, a total of 771 Kemp's ridley nests have been documented on the Texas coast. This is more than nine times greater than the 81 nests recorded over the previous 54 years from 1948-2001, indicating an increasing nesting population in Texas. From 2005 through 2009, the number of nests from all monitored beaches indicate approximately 5,500 females are nesting each season in the Gulf of Mexico (NMFS and USFWS 2010b).

The Turtle Expert Working Group (2000) estimated that the population size of Kemp's ridley sea turtles grew at an average rate of l1.3 percent per year (95% C.I. slope = 0.096-0.130) between 1985 and 1998. Over the same time interval, hatchling production increased at a slightly slower rate (9.5% per year). Population models predict the population will grow 12-16% per year, for the near future, assuming current survival rates within each life stage remain constant (Heppell *et al*. 2005 *in* NMFS 2010b).

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

The number of adult males in the population is unknown, but sex ratios of hatchlings and immature Kemp's ridleys suggest that the population is female-biased, suggesting that the number of adult males is less than the number of adult females (NMFS and USFWS 2007c). While there is cautious optimism for recovery, events such as the Deepwater Horizon oil release, and stranding events associated with increased skimmer trawl use and poor TED compliance in the northern Gulf of Mexico may dampen recent population growth.

As with the other sea turtle species, fishery mortality accounts for a large proportion of annual human-caused mortality outside the nesting beaches, while other activities like dredging, pollution, and habitat destruction also contribute to annual human caused mortality, but the levels are unknown. Based on their five-year status review of the species, NMFS and USFWS (2007c) determined that Kemp's ridley sea turtles should not be reclassified as threatened under the ESA. A revised bi-national recovery plan was published for public comment in 2010, and in September 2011, NMFS, USFWS, and the Services and the Secretary of Environment and Natural Resources, Mexico (SEMARNAT) released the second revision to the Kemp's ridley recovery plan. Based on the information presented above, for purposes of this Opinion, we consider that the status of Kemp's ridleys over the next ten years will be no worse than it is currently and that the species may actually be in the early stages of recovery, although this should be viewed in the context of a much larger population in the mid-20th century.

Hearing

There is no information on Kemp's ridley sea turtle hearing. However, we assume that their hearing sensitivities will be similar to those of green and loggerhead sea turtles with their best hearing sensitivity in the low frequency range: from 50 to 400 Hz with rapid declines for tones at lower and higher frequencies and a practical upper limit of about 1600 Hz in water (Piniak *et al*. 2012).

Leatherback Sea Turtle

Distribution

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

Leatherback sea turtles are highly migratory, exploiting convergence zones and upwelling areas in the open ocean, along continental margins, and in archipelagic waters (Morreale *et al.* 1994, Eckert 1998, Eckert 1999). In a single year, a leatherback may swim more than 10,000 kilometers (Eckert 1998). In the North Atlantic Ocean, leatherback turtles regularly occur in deep waters (>328 ft). An aerial survey in the north Atlantic observed leatherback turtles in continental shelf and pelagic environments with sightings in offshore waters ranging from 7-27^{o} C (CETAP 1982). Juvenile leatherbacks usually stay in warmer, tropical waters >21^{o} C (Eckert 2002). Males and females show some degree of natal homing to annual breeding sites (James et al. 2005).

In the Pacific Ocean, leatherback turtles have been reported in all pelagic waters of the Pacific between 71°N and 47°S latitude and in all other major pelagic ocean habitats (NMFS and USFWS 1998c). Leatherback turtles lead a completely pelagic existence, foraging widely in temperate waters except during the nesting season, when gravid females return to tropical beaches to lay eggs. Males are rarely observed near nesting areas, and it has been hypothesized that leatherback sea turtles probably mate outside of tropical waters, before females swim to their nesting beaches (Eckert and Eckert 1988).

Population Structure

Leatherback sea turtles are widely distributed throughout the oceans of the world, including the Atlantic, Pacific, and Indian Oceans, and the Mediterranean Sea (Casale et al. 2003, Ernst and Barbour 1972, Hamann et al. 2006b), although they are not known to nest there. These four main regional areas may further be divided into nesting aggregations. Leatherback turtles are found on the western and eastern coasts of the Pacific Ocean, with nesting aggregations in Mexico and Costa Rica (eastern Pacific) and Malaysia, Indonesia, Australia, the Solomon Islands, Papua New Guinea, Thailand, and Fiji (western Pacific). In the Atlantic Ocean, leatherback nesting aggregations have been documented in Gabon, Sao Tome and Principe, French Guiana, Suriname, and Florida. In the Caribbean, leatherbacks nest in the U.S. Virgin Islands and Puerto Rico. In the Indian Ocean, leatherback nesting aggregations are reported in India and Sri Lanka, and the Andaman and Nicobar Islands. Detailed population structure is unknown, but is likely dependent upon nesting beach location.

Atlantic Ocean

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

Caribbean Sea

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

Mediterranean Sea

Casale et al. (2003) reviewed the distribution of leatherback sea turtles in the Mediterranean. Among the 411 individual records of leatherback sightings in the Mediterranean, there were no nesting records. Nesting in the Mediterranean is believed to be extremely rare, if it occurs at all. Leatherbacks found in Mediterranean waters originate from the Atlantic Ocean.

Indian Ocean

Leatherbacks nest in several areas around the Indian Ocean. These sites include Tongaland, South Africa (Pritchard 2002) and the Andaman and Nicobar Islands (Andrews et al. 2002, Hamann *et al*. 2006b). Intensive survey and tagging work in 2001 provided new information on the level of nesting in the Andaman and Nicobar Islands (Andrews et al. 2002). Based on the survey and tagging work, it was estimated that 400-500 female leatherbacks nest annually on Great Nicobar Island (Andrews et al. 2002). The number of nesting females using the Andaman and Nicobar Islands combined was estimated to be around 1,000 (Andrews and Shanker 2002). Some nesting also occurs along the coast of Sri Lanka, although in much smaller numbers than in the past (Pritchard 2002).

Pacific Ocean

Leatherbacks are found from tropical waters north to Alaska within the North Pacific and is the most common sea turtle in the eastern Pacific north of Mexico (Eckert, 1993; Stinson, 1984; Wing & Hodge, 2002). The west coast of Central America and Mexico hosts nesting from September-March, although Costa Rican nesting peaks during April-May (Chacón-Chaverri & Eckert, 2007; LGL Ltd., 2007). Leatherback nesting aggregations occur widely in the Pacific, including China, Malaysia, Papua New Guinea, Indonesia, Thailand, Australia, Fiji, the Solomon Islands, and Central America (P. H. Dutton et al., 2007; Limpus, 2002). Significant nesting also occurs along the Central American coast (Márquez, 1990). Although not generally known to nest on Japanese shores, two nests were identified in the central Ryukyu Islands in 2002 (Kamezaki, Oki, Mizuno, Toji, & Doi, 2002). Leatherback nesting has been declining at all major Pacific Basin nesting beaches for the last two decades (Spotila et al. 1996, 2000; NMFS and USFWS 1998a, 2007b; Sarti et al. 2000).

In the Pacific, nesting beaches in Mexico and Costa Rica (nesting occurs October through March) are a separate population from the western Pacific beaches (Benson et al., 2007; summary in NMFS and USFWS, 2007d; J. R. Spotila, 2004). In Costa Rica, leatherbacks nest at Playa Naranjo in Santa Rosa National Park, the second-most important nesting beach on the Pacific coast (Yañez, Gaos, & Aruaz, 2010), Rio Oro on the Osa Peninsula, and at various beaches in Las Baulas National Park, which includes Playa Langosta and Playa Grande and contains the largest colony of leatherbacks in the Pacific (J. R. Spotila, 2004). Females typically lay six clutches per season (average nine days between nests), which incubate for 58–65 days (Lux, Reina, & Stokes, 2003). Limited nesting also occurs along Nicaragua, Panama, El Salvador, and Guatemala.). A dramatic decline has been seen on nesting beaches in Pacific Mexico, where aerial survey data was used to estimate that tens of thousands of leatherback nests were laid on the beaches in the 1980s (Pritchard 1982), but a total of only 120 nests on the four primary index beaches (combined) were counted in the 2003-2004 season (Sarti Martinez et al. 2007). Since the early

1980s, the Mexican Pacific population of adult female leatherback turtles has declined to slightly more than 200 during 1998-1999 and 1999-2000 (Sarti et al. 2000). Spotila et al. (2000) reported the decline of the leatherback nesting at Playa Grande, Costa Rica, which had been the fourth largest nesting group in the world and the most important nesting beach in the Pacific.

Threats to the Species

Natural Threats. The various habitat types leatherback sea turtles occupy throughout their lives exposes these sea turtles to a wide variety of natural threats. The beaches on which leatherback sea turtles nest and the nests themselves are threatened by hurricanes and tropical storms as well as the storm surges, sand accretion, and rainfall that are associated with hurricanes. Hatchlings are hunted by predators like herons, gulls, dogfish, and sharks. Larger leatherback sea turtles, including adults, are also killed by sharks and other large, marine predators.

Anthropogenic Threats. Leatherback sea turtles are endangered by several human activities, including fisheries interactions, entanglement in fishing gear (e.g., gillnets, longlines, lobster pots, weirs), direct harvest, egg collection, the destruction and degradation of nesting and coastal habitat, boat collisions, and ingestion of marine debris (NMFS and USFWS 2007, TEWG 2007).

The foremost threat is the number of leatherback turtles killed or injured in fisheries (Finkbeiner 2011). Leatherbacks seem to be the most vulnerable to entanglement in fishing gear, particularly trap/pot gear. This susceptibility may be the result of their body type (large size, long pectoral flippers, and lack of a hard shell), their diving and foraging behavior, their distributional overlap with the gear, their possible attraction to gelatinous organisms and algae that collect on buoys and buoy lines at or near the surface, and perhaps to the lightsticks used to attract target species in longline fisheries. Leatherbacks entangled in fishing gear generally have a reduced ability to feed, dive, surface to breathe, or perform any other behavior essential to survival (Balazs 1985). In addition to drowning from forced submergence, they may be more susceptible to boat strikes if forced to remain at the surface, and entangling lines can constrict blood flow resulting in tissue necrosis. The long-term impacts of entanglement on leatherback health remain unclear.

Spotila (2000) concluded that a conservative estimate of annual leatherback fishery-related mortality (from longlines, trawls and gillnets) in the Pacific Ocean during the 1990s is 1,500 animals. He estimates that this represented about a 23% mortality rate (or 33% if most mortality was focused on the East Pacific population). Spotila (2000) asserts that most of the mortality associated with the Playa Grande nesting site was fishery related.

Leatherback sea turtles are exposed to commercial fisheries in many areas of the Atlantic Ocean. For example, leatherback entanglements in fishing gear are common in Canadian waters where Goff and Lien (1988) reported that 14 of 20 leatherbacks encountered off the coast of Newfoundland and Labrador were entangled in fishing gear including salmon net, herring net, gillnet, trawl line and crab pot line. Leatherbacks are reported taken by the many other nations that participate in Atlantic pelagic longline fisheries (see NMFS 2001, for a complete description of take records), including Taiwan, Brazil, Trinidad, Morocco, Cyprus, Venezuela, Korea, Mexico, Cuba, U.K., Bermuda, People's Republic of China, Grenada, Canada, Belize, France, and Ireland.

In the Pacific Ocean, between 1,000 and 1,300 leatherback sea turtles are estimated to have been captured and killed in longline fisheries in 2000 (Lewison *et al.* 2004). Shallow-set longline fisheries based out of Hawai'i are estimated to have captured and killed several hundred leatherback sea turtles before they were closed in 2001. When they were re-opened in 2004, with substantial modifications to protect sea turtles, these fisheries were estimated to have captured and killed about 1 or 2 leatherback sea turtles each year. Between 2004 and 2008, shallow-set fisheries based out of Hawai'i are estimated to have captured about 19 leatherback sea turtles, killing about 5 of these sea turtles. A recent biological opinion on these fisheries expected this rate of interaction and deaths to continue into the foreseeable future (NMFS 2012). Leatherback sea turtles have also been and are expected to continue to be captured and killed in the deep-set based longline fisheries based out of Hawai'i and American Samoa.

Leatherback interactions with the U.S. South Atlantic and Gulf of Mexico shrimp fisheries are also known to occur (NMFS 2002a). Leatherbacks are likely to encounter shrimp trawls working in the coastal waters off the U.S. Atlantic coast (from Cape Canaveral, Florida through North Carolina) as they make their annual spring migration north. For many years, TEDs that were required for use in the U.S. South Atlantic and Gulf of Mexico shrimp fisheries were less effective for leatherbacks as compared to the smaller, hard-shelled turtle species, because the TED openings were too small to allow leatherbacks to escape. To address this problem, NMFS issued a final rule on February 21, 2003, to amend the TED regulations (68 FR 8456, February 21, 2003). Modifications to the design of TEDs are now required in order to exclude leatherbacks as well as large benthic immature and sexually mature loggerhead and green sea turtles. Given those modifications, Epperly et al. (2002) anticipated an average of 80 leatherback mortalities a year in shrimp gear interactions but dropping to an estimate of 26 leatherback mortalities in 2009 due to effort reduction in the Southeast shrimp fishery (Memo from Dr. B. Ponwith, SEFSC, to Dr. R. Crabtree, SERO; January 5, 2011).

Along the Atlantic coast of the U.S., NMFS estimated that about 800 leatherback sea turtles are captured in pelagic longline fisheries, bottom longline and drift gillnet fisheries for sharks as well as lobster, deep-sea red crab, Jonah crab, dolphin fish and wahoo, and Pamlico Sound gillnet fisheries. For instance, an estimated 6,363 leatherback sea turtles were caught by the U.S. Atlantic tuna and swordfish longline fisheries between 1992 and 1999 (SEFSC 2001). 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 three-year period starting in 2007 (NMFS 2004c). In 2010, there were 26 observed interactions between leatherback sea turtles and longline gear used in the HMS fishery (Garrison and Stokes 2012). All leatherbacks were released alive, with all gear removed in 14 (53.8%) of the 26 captures. A total of 171 (95% CI: 104.3-280.2) leatherback sea turtles are estimated to have interacted with the longline fisheries managed under the HMS FMP in 2010 based on the observed bycatch events (Garrison and Stokes 2012). The 2010 estimate continues a downward trend since 2007 and remains well below the average prior to implementation of gear regulations (Garrison and Stokes 2012).

Leatherback sea turtles are known to drown in fish nets set in coastal waters of Sao Tome, West Africa (Castroviejo *et al.* 1994; Graff 1995). Gillnets are one of the suspected causes for the decline in the leatherback turtle population in French Guiana (Chevalier *et al.* 1999), and gillnets targeting green and hawksbill turtles in the waters of coastal Nicaragua also incidentally catch leatherback turtles (Lagueux *et al.* 1998). Observers on shrimp trawlers operating in the northeastern region of Venezuela documented the capture of six leatherbacks from 13,600 trawls (Marcano and Alio, 2000). An estimated 1,000 mature female leatherback turtles are caught annually off of Trinidad and Tobago with mortality estimated to be between 50-95% (Eckert and Lien, 1999). However, many of the turtles do not die as a result of drowning, but rather because the fishermen butcher them in order to get them out of their nets (NMFS 2001). There are known to be many sizeable populations of leatherbacks nesting in West Africa, possibly as many as 20,000 females nesting annually (Fretey 2001). In Ghana, nearly two thirds of the leatherback turtles that come up to nest on the beach are killed by local fishermen.

On some beaches, nearly 100% of the eggs laid have been harvested. Eckert (1996) and Spotila *et al.* (1996) noted that adult mortality has also increased significantly, particularly as a result of driftnet and longline fisheries. Like green and hawksbill sea turtles, leatherback sea turtles are threatened by domestic or domesticated animals that prey on their nests; artificial lighting that disorients adult female and hatchling sea turtles, which can dramatically increase the mortality rates of hatchling sea turtles; beach replenishment; ingestion and entanglement in marine debris; and environmental contaminants.

Leatherbacks may be more susceptible to marine debris ingestion than other sea turtle species due to the tendency of floating debris to concentrate in convergence zones that juveniles and adults use for feeding (Shoop and Kenney 1992; Lutcavage et al. 1997). Investigations of the necropsy results of leatherback sea turtles revealed that a substantial percentage (34% of the 408 leatherback necropsies recorded between 1885 and 2007) reported plastic within the turtle's stomach contents, and in some cases (8.7% of those cases in which plastic was reported), blockage of the gut was found in a manner that may have caused the mortality (Mrosovsky et al. 2009). An increase in reports of plastic ingestion was evident in leatherback necropsies conducted after the late 1960s (Mrosovsky et al. 2009). Along the coast of Peru, intestinal contents of 19 of 140 (13%) leatherback carcasses were found to contain plastic bags and film (Fritts 1982). The presence of plastic debris in the digestive tract suggests that leatherbacks might not be able to distinguish between prey items (e.g., jellyfish) and plastic debris (Mrosovsky 1981). Balazs (1985) speculated that plastic objects may resemble food items by their shape, color, size, or even movements as they drift about, and induce a feeding response in leatherbacks.

Status

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. In 1979, critical habitat for leatherback turtles was designated to include the coastal waters adjacent to Sandy Point, St. Croix, U.S. Virgin Islands. On September 26, 2007, NMFS received a petition to revise the critical habitat designation for leatherback sea turtles to include waters along the U.S. West Coast. On January 26, 2012, NMFS published a final rule to revise the critical habitat designation to include three particular areas of marine habitat. The designation includes approximately 16,910 square miles along the California coast from Point Arena to Point Arguello east of the 3,000 meter depth contour, and 25,004 square miles from Cape Flattery, Washington to Cape Blanco, Oregon east of the 2,000 meter depth contour. The areas comprise approximately 41,914 square miles of marine habitat and include waters from the ocean surface down to a maximum depth of 262 feet. The designated critical habitat areas contain the physical or biological feature essential to the conservation of the species that may require special management conservation or protection. In particular, the team identified one Primary Constituent Element: the occurrence of prey species, primarily scyphomedusae of the order Semaeostomeae, of sufficient condition, distribution, diversity, abundance and density necessary to support individual as well as population growth, reproduction, and development of leatherbacks.

Globally, leatherback turtle populations have been decimated worldwide. Increases in the number of nesting females have been noted at some sites in the Atlantic Ocean, but these are far outweighed by local extinctions, especially of island populations, and the demise of populations throughout the Pacific, such as in Malaysia and Mexico. Spotila et al. (1996) estimated the global population of female leatherback turtles to be only 34,500 (confidence limits: 26,200 to 42,900), but later issued an update of 35,860 (Spotila 2004b). However, the eastern Pacific population has continued to decline since that estimate, leading some researchers to conclude that the leatherback is now on the verge of extinction in the Pacific Ocean (e.g. Spotila et al. 1996, Spotila, et al. 2000). The species as a whole is declining and local populations are in danger of extinction (NMFS 2001b; NMFS 2001a).

In the Pacific Ocean, leatherback sea turtles are critically endangered as a direct consequence of a historical combination of overexploitation and habitat loss. The information available suggests that leatherback sea turtles have high probabilities of becoming extinct in the Pacific Ocean unless they are protected from the combined threats of entanglements in fishing gear, overharvests, and loss of their nesting habitat. The limited data available suggests that leatherback sea turtles exist at population sizes small enough to be classified as “small” populations (that is, populations that exhibit population dynamics that increase the extinction probabilities of the species or several of its populations) as evidenced by biases in the male to female ratios in the Pacific.

Populations in the eastern Atlantic (i.e. off Africa) and Caribbean appear to be stable; however, information regarding the status of the entire leatherback population in the Atlantic is lacking and it is certain that some nesting populations (e.g., St. John and St. Thomas, U.S. Virgin Islands) have been extirpated (NMFS and USFWS 1995). Data collected in southeast Florida clearly indicate increasing numbers of nests for the past twenty years (9.1-11.5% increase), although it is critical to note that there was also an increase in the survey area in Florida over time (NMFS 2001). However, the largest leatherback rookery in the western North Atlantic remains along the northern coast of South America in French Guiana and Suriname. Recent information suggests that Western Atlantic populations declined from 18,800 nesting females in 1996 (Spotila et al. 1996) to 15,000 nesting females by 2000 (Spotila, personal communication cited in NMFS 2001). The nesting population of leatherback turtles in the Suriname-French Guiana trans-boundary region has been declining since 1992 (Chevalier and Girondot, 1998). Poaching and fishing gear interactions are believed to be the major contributors to the decline of leatherbacks in the area. The status of leatherback sea turtles in the Atlantic Ocean remains uncertain.

Diving Behavior

The maximum dive depths for post-nesting female leatherback turtles in the Caribbean have been recorded at 475 meters and over 1,200 meters, with routine dives recorded at between 50 and 84 meters (Bradshaw et al., 2007; Fossette et al. 2010, Houghton et al., 2008). The maximum dive length recorded for such female leatherback turtles was 37.4 minutes, while routine dives ranged from 4 -14.5 minutes (in Lutcavage and Lutz 1997). Leatherback turtles also appear to spend almost the entire portion of each dive traveling to and from maximum depth, suggesting that maximum exploitation of the water column is of paramount importance to the leatherback (Eckert et al. 1989).

A total of six adult female leatherback turtles from Playa Grande, Costa Rica were monitored at sea during their internesting intervals and during the 1995 through 1998 nesting seasons. The turtles dived continuously for the majority of their time at sea, spending 57 - 68% of their time submerged. Mean dive depth was 19±1 meters and the mean dive duration was 7.4± 0.6 minutes (Southwood et al. 1999). Similarly, Eckert (1999) placed transmitters on nine leatherback females nesting at Mexiquillo Beach and recorded dive behavior during the nesting season. The majority of the dives were less than 150 meters depth, although maximum depths ranged from 132 meters to over 750 meters. Although the dive durations varied between individuals, the majority of them made a large proportion of very short dives (less than two minutes), although Eckert (1999) speculates that these short duration dives most likely represent just surfacing activity after each dive. Excluding these short dives, five of the turtles had dive durations greater than 24 minutes, while three others had dive durations between 12 - 16 minutes.

Migrating leatherback turtles also spend a majority of time at sea submerged, and they display a pattern of continual diving (Standora et al. 1984, in Southwood et al. 1999). Based on depth profiles of four leatherbacks tagged and tracked from Monterey Bay, California in 2000 and 2001, using satellite-linked dive recorders, most of the dives were to depths of less than 100 meters and most of the time was spent shallower than 80 meters. Based on preliminary analyses of the data, 75-90% of the time the leatherback turtles were at depths less than 80 meters.

Hearing

There is no information on leatherback sea turtle hearing. However, we assume that their hearing sensitivities will be similar to those of green and loggerhead sea turtles with their best hearing sensitivity in the low frequency range: from 50 to 400 Hz with rapid declines for tones at lower and higher frequencies and a practical upper limit of about 1600 Hz in water (Piniak *et al*. 2012).

Loggerhead Sea Turtle

Distribution

Loggerheads are circumglobal, inhabiting continental shelves, bays, estuaries, and lagoons throughout the temperate and tropical regions of the Atlantic, Pacific, and Indian Oceans (Dodd 1988). However, the majority of loggerhead nesting is at the western rims of the Atlantic and Indian Oceans. The most recent reviews show that only two loggerhead nesting aggregations have greater than 10,000 females nesting per year: Peninsular Florida, United States and Masirah Island, Oman (Baldwin et al. 2003, Ehrhart et al. 2003, Kamezaki et al. 2003, Limpus and Limpus 2003b, Margaritoulis et al. 2003). Nesting aggregations with 1,000 to 9,999 females nesting annually are Georgia through North Carolina (U.S.), Quintana Roo and Yucatán (Mexico), Brazil, Cape Verde Islands (Cape Verde), Western Australia (Australia), and Japan. Smaller nesting aggregations with 100 to 999 nesting females annually occur in the Northern Gulf of Mexico (U.S.), Dry Tortugas (U.S.), Cay Sal Bank (The Bahamas), Tongaland (South Africa), Mozambique, Arabian Sea Coast (Oman), Halaniyat Islands (Oman), Cyprus, Peloponnesus (Greece), Zakynthos (Greece), Crete (Greece), Turkey, and Queensland (Australia). In contrast to determining population size on nesting beaches, determining population size in the marine environment has been very localized (Bjorndal and Bolten 2000). At present, there are no data on population size in the oceanic habitat.

Adult loggerheads are known to make considerable migrations from nesting beaches to foraging grounds (TEWG, 1998); and evidence indicates turtles entering the benthic environment undertake routine migrations along the coast that are limited by seasonal water temperatures. Small juveniles are found in pelagic waters (e.g., of the North Atlantic and the Mediterranean Sea); and the transition from oceanic to neritic juvenile stages can involve trans-oceanic migrations (Bowen et al., 2004). Loggerhead nesting is confined to lower latitudes, concentrated in temperate zones and subtropics; the species generally does not nest in tropical areas (NMFS and USFWS, 1991b; NRC, 1990; Witherington, Herren, & Bresette, 2006b). Loggerhead turtles travel to northern waters during spring and summer as water temperatures warm, and southward and offshore toward warmer waters in fall and winter; loggerheads are noted to occur year round in offshore waters of sufficient temperature.

Ehrhart *et al.* (2003) provided a summary of the literature identifying known nesting habitats and foraging areas for loggerheads within the Atlantic Ocean. Detailed information is also provided in the five-year status review for loggerheads (NMFS and USFWS 2007a), the TEWG report (2009), and the final revised recovery plan for loggerheads in the Northwest Atlantic Ocean (NMFS and USFWS 2009), which is a second revision to the original recovery plan that was approved in 1984 and subsequently revised in 1991.

In the western Atlantic, waters as far north as 41° N to 42° N latitude are foraging habitats for juveniles, as well as adults (Shoop 1987; Shoop and Kenney 1992; Ehrhart *et al.* 2003; Mitchell *et al.* 2003). In U.S. Atlantic waters, loggerheads commonly occur throughout the inner continental shelf from Florida to Cape Cod, Massachusetts and in the Gulf of Mexico from Florida to Texas, although their presence varies with the seasons due to changes in water temperature (Shoop and Kenney 1992; Epperly *et al.* 1995a, 1995b; Braun and Epperly 1996; Braun-McNeill *et al.* 2008; Mitchell *et al.* 2003). Loggerheads have been observed in waters with surface temperatures of 7 C to 30 C, but water temperatures ≥11 C are most favorable (Shoop and Kenney 1992; Epperly *et al.* 1995b). The presence of loggerhead sea turtles in U.S. Atlantic waters is also influenced by water depth. Aerial surveys of continental shelf waters north of Cape Hatteras, North Carolina indicated that loggerhead sea turtles were most commonly sighted in waters with bottom depths ranging from 22-49 meters deep (Shoop and Kenney 1992). However, more recent survey and satellite tracking data support that they occur in waters from the beach to beyond the continental shelf (Mitchell *et al.* 2003; Braun-McNeill and Epperly 2004; Mansfield 2006; Blumenthal *et al.* 2006; Hawkes *et al.* 2006, 2011; McClellan and Read 2007; Mansfield *et al*. 2009).

Loggerhead sea turtles occur year round in ocean waters off North Carolina, South Carolina, Georgia, and Florida. In these areas of the South Atlantic Bight, water temperature is influenced by the proximity of the Gulf Stream. As coastal water temperatures warm in the spring, loggerheads begin to migrate to inshore waters of the Southeast United States (*e.g.*, Pamlico and Core Sounds) and also move up the U.S. Atlantic Coast (Epperly *et al.* 1995a, 1995b, 1995c; Braun-McNeill and Epperly 2004), occurring in Virginia foraging areas as early as April/May and on the most northern foraging grounds in the Gulf of Maine in June (Shoop and Kenney 1992). The trend is reversed in the fall as water temperatures cool. The large majority leave the Gulf of Maine by mid-September but some turtles may remain in Mid-Atlantic and Northeast areas until late fall. By December, loggerheads have migrated from inshore and more northern coastal waters to waters offshore of North Carolina, particularly off of Cape Hatteras, and waters further south where the influence of the Gulf Stream provides temperatures favorable to sea turtles (Shoop and Kenney 1992; Epperly *et al.* 1995b).

Population Structure

Loggerhead sea turtles, like other sea turtles, are divided into regional groupings that represent major oceans or seas: the Atlantic Ocean, Pacific Ocean, Indian Ocean, Caribbean Sea and Mediterranean Sea.

In these regions, the population structure of loggerhead turtles is based on physical, ecological, behavioral and genetic data and distribution of their nesting aggregations.

Recent studies have established that the loggerhead's life history is more complex than previously believed. Rather than making discrete developmental shifts from oceanic to neritic environments, research is showing that both adults and (presumed) neritic stage juveniles continue to use the oceanic environment and will move back and forth between the two habitats (Witzell 2002; Blumenthal *et al.* 2006; Hawkes *et al.* 2006; McClellan and Read 2007; Mansfield *et al*. 2009). One of the studies tracked the movements of adult post-nesting females and found that differences in habitat use were related to body size with larger adults staying in coastal waters and smaller adults traveling to oceanic waters (Hawkes *et al.* 2006). A tracking study of large juveniles found that the habitat preferences of this life stage were also diverse with some remaining in neritic waters and others moving off into oceanic waters (McClellan and Read 2007). However, unlike the Hawkes *et al.* (2006) study, there was no significant difference in the body size of turtles that remained in neritic waters versus oceanic waters (McClellan and Read 2007).

Pelagic and benthic juveniles are omnivorous and forage on crabs, mollusks, jellyfish, and vegetation at or near the surface (Dodd 1988; NMFS and USFWS 2008). Sub-adult and adult loggerheads are primarily coastal dwelling and typically prey on benthic invertebrates such as mollusks and decapod crustaceans in hard bottom habitats (NMFS and USFWS 2008). For the past decade or so, the scientific literature has recognized five distinct nesting groups, or subpopulations, of loggerhead sea turtles in the Northwest Atlantic, divided geographically as follows: (1) a northern group of nesting females that nest from North Carolina to northeast Florida at about 29 N latitude; (2) a south Florida group of nesting females that nest from 29 N latitude on the East Coast to Sarasota on the West Coast; (3) a Florida Panhandle group of nesting females that nest around Eglin Air Force Base and the beaches near Panama City, Florida; (4) a Yucatán group of nesting females that nest on beaches of the eastern Yucatán Peninsula, Mexico; and (5) a Dry Tortugas group that nests on beaches of the islands of the Dry Tortugas, near Key West, Florida and on Cal Sal Bank (TEWG 2009).

The lack of genetic structure makes it difficult to designate specific boundaries for the nesting subpopulations based on genetic differences alone. Therefore, the Status Review Team (Conant et al 2009) recently used a combination of geographic distribution of nesting densities, geographic separation, and geopolitical boundaries, in addition to genetic differences, to reassess the designation of these subpopulations to identify recovery units. Five recovery units have been designated for the Northwest Atlantic population of loggerhead sea turtles based on the aforementioned nesting groups and inclusive of a few other nesting areas not mentioned above. The first four of these recovery units represent nesting assemblages located in the Southeast United States. The fifth recovery unit is composed of all other nesting assemblages of loggerheads within the Greater Caribbean, outside the United States, but which occur within U.S. waters during some portion of their lives. The five recovery units representing nesting assemblages are: (1) the Northern Recovery Unit (NRU: Florida/Georgia border through southern Virginia), (2) the Peninsular Florida Recovery Unit (PFRU: Florida/Georgia border through Pinellas County, Florida), (3) the Dry Tortugas Recovery Unit (DTRU: islands located west of Key West, Florida), (4) the Northern Gulf of Mexico Recovery Unit (NGMRU: Franklin County, Florida through Texas), and (5) the Greater Caribbean Recovery Unit (GCRU: Mexico through French Guiana, Bahamas, Lesser Antilles, and Greater Antilles)(NMFS and USFWS 2008).

Genetic studies of juvenile and a few adult loggerhead sea turtles collected from Northwest Atlantic foraging areas (beach strandings, a power plant in Florida, and North Carolina fisheries) show that the loggerheads that occupy East Coast U.S. waters originate from these Northwest Atlantic nesting groups; primarily from the nearby nesting beaches of southern Florida, as well as the northern Florida to North Carolina beaches, and finally from the beaches of the Yucatán Peninsula, Mexico (Rankin-Baransky *et al*. 2001; Witzell *et al*. 2002; Bass *et al.* 2004; Bowen *et al*. 2004). The contribution of these three nesting assemblages varies somewhat among the foraging habitats and age classes surveyed along the east coast.

Threats to the Species

Natural Threats. The diversity of a loggerhead sea turtle's life history and the various habitat types loggerhead sea turtles occupy throughout their lives exposes these sea turtles to a wide variety of natural and anthropogenic threats. The five-year status review and 2009 recovery plan provide a summary of natural as well as anthropogenic threats to loggerhead sea turtles (NMFS and USFWS 2007a, 2009). Amongst those of natural origin, hurricanes are known to be destructive to sea turtle nests (Witherington et al. 2006). Sand accretion, rainfall, and wave action that result from these storms can appreciably reduce hatchling success. Other sources of natural mortality include cold-stunning (Witherington and Ehrhart 1989, Morreale et al. 1992), and native species predation such as herons, gulls, dogfish, sharks and other large marine predators. Biotoxin exposure by harmful algal blooms, such as a red tide, also impact loggerheads. *Karenia brevis*, a type of microalgae known as a dinoflagellate that produces a toxin, has been known to cause red tides (Florida Marine Research Institute 2003).

Anthropogenic Threats. A wide variety of anthropogenic activities adversely affect hatchlings and adult female turtles and their nesting habitat include coastal development/construction of fishing piers that alter patterns of erosion and accretion on nesting beaches, placement of erosion control structures and other barriers to nesting, beachfront lighting, vehicular and pedestrian traffic, sand extraction, beach erosion, beach sand placement, beach pollution (ingestion and entanglement of debris and environmental contaminants), removal of native vegetation, and poaching (Baldwin 1992; NMFS and FWS 1998, 2008; Margaritoulis et al. 2003). As the size of the human population in coastal areas increases, that population brings with it secondary threats such as exotic fire ants, feral hogs, dogs, and the growth of populations of native species that tolerate human presence (*e.g.*, raccoons, armadillos, and opossums) and which feed on turtle eggs (NMFS and USFWS 2007a, 2009). Planting of non-native plants can have a detrimental effect in the form of roots invading eggs (e.g., tamarisk tree roots invading eggs in Zakynthos, Greece) (Margaritoulis et al. 2007).

When they are in coastal or marine waters, loggerhead turtles are affected by a completely different set of human activities that include discharges of toxic chemicals and other pollutants into the marine ecosystem; underwater explosions; hopper dredging, offshore artificial lighting; entrainment or impingement in power plants; entanglement in marine debris; ingestion of marine debris; boat collisions; poaching, and interactions with commercial fisheries. Human presence, especially in neritic habitats, can disturb both juvenile and adult loggerheads. The ever-growing human population in coastal areas and the corresponding increase in recreational boating, and diving can result in behavioral disturbances to resting, foraging, and migrating loggerheads. Continuous, intense boat traffic in neritic habitats may result in abandonment of previously used foraging or resting areas, and intense diver/snorkeler activity may result in displacement of loggerheads from preferred resting or foraging areas (NMFS 2009 Recovery Plan).

Interactions with fisheries represent a primary threat because of the number of individuals that are captured and killed in fishing gear each year (Finkbeiner et al. 2011). In the Pacific Ocean, between 2,600 and 6,000 loggerhead sea turtles are estimated to have been captured and killed in longline fisheries in 2000 (Lewison *et al.* 2004). Shallow-set Hawai'i based longline fisheries are estimated to have captured and killed several hundred loggerhead sea turtles before they were closed in 2001. When they were re-opened in 2004, with substantial modifications to protect sea turtles, these fisheries were estimated to have captured and killed about fewer than 5 loggerhead sea turtles each year. Between 2004 and 2008, shallow-set fisheries based out of Hawai'i are estimated to have captured about 45 loggerhead sea turtles, killing about 10 of these sea turtles. A recent biological opinion on these fisheries expected this rate of interaction and deaths to continue into the foreseeable future (NMFS 2008). Loggerhead sea turtles have also been and are expected to continue to be captured and killed in the deep-set based longline fisheries based out of Hawai'i and American Samoa.

Shrimp trawl fisheries account for the highest number of loggerhead sea turtles that are captured and killed, but they are also captured and killed in trawls, traps and pots, longlines, and dredges along the Atlantic coast of the U.S. In 2002, NMFS estimated that almost 163,000 loggerhead sea turtles are captured in shrimp trawl fisheries each year in the Gulf of Mexico, with 3,948 of those sea turtles dying as a result of their capture. Each year, several hundred loggerhead sea turtles are also captured in herring fisheries; mackerel, squid, and butterfish fisheries; monkfish fisheries; pound net fisheries, summer flounder and scup fisheries; Atlantic pelagic longline fisheries; and gillnet fisheries in Pamlico Sound. Although most of these turtles are released alive, these fisheries capture about 2,000 loggerhead sea turtles each year, killing almost 700; the effects of capture-related stress on the current or expected future reproductive success of sea turtles remains unknown.

In the pelagic environment, loggerhead sea turtles are exposed to a series of longline fisheries that include the U.S. Atlantic tuna and swordfish longline fisheries, an Azorean longline fleet, a Spanish longline fleet, and various fleets in the Mediterranean Sea (Aguilar *et al.* 1995, Bolten *et al.* 1994, Crouse 1999). In the benthic environment in waters off the coastal U.S., loggerheads are exposed to a suite of fisheries in federal and state waters including trawl, purse seine, hook and line, gillnet, pound net, longline, dredge, and trap fisheries.

Status

Northwest Atlantic loggerhead sea turtles were listed as threatened on October 24, 2011 (76 FR 58868, September 22, 2011).

Atlantic Ocean

Loggerheads in the Northwest Atlantic Ocean comprise one of the two largest nesting assemblages in the world and have been identified as the most significant assemblage in the western hemisphere. Data collected from 1989 to 2010 indicates that there has been a slight decline in nesting numbers but the decline is not statistically different from zero (76 FR 58899, September 22, 2011). Although this DPS suffers impacts from U.S. fisheries and degradation of its terrestrial and marine habitats along their range in the Atlantic and Gulf of Mexico, the nesting population is large and widespread and trends indicate that at least the nesting population appears to be stabilizing, as substantial conservation efforts are underway to address threats to this species’ survival and recovery. Five nesting subpopulations have been identified in the Northwest Atlantic Ocean (NMFS and USFWS 2008). Their status follows:

(1) Northern U.S. (Florida/Georgia border to southern Virginia). The Northern U.S. subpopulation is the second largest unit within the Northwest Atlantic population and has been declining at 1.3 percent annually from 1983 to 2007 (NMFS and USFWS, 2008). Based on more recent nesting data from Georgia, South Carolina and North Carolina from 2008 through 2010, however, this subpopulation may be showing signs of stabilizing (76 FR 58885, September 22, 2011);

(2) Peninsular Florida (Florida/Georgia border south through Pinellas County, excluding the islands west of Key West, Florida). A significant declining trend had been documented for the Peninsular Florida subpopulation, where nesting declined 26 percent over the 20-year period from 1989–2008, and declined 41 percent over the period 1998–2008 (NMFS and USFWS, 2008; Witherington *et al.*, 2009). Extending the nesting data set from 1989 through the 2010 nesting season yields a slightly declining trend not significant from zero. Nesting in 2008 substantially surpassed nesting in 2007 and nesting in 2010 reached the highest levels since 2000. This subpopulation represents approximately 87 percent of all nesting effort in the Northwest Atlantic Ocean DPS (Ehrhart *et al.*, 2003);

(3) Dry Tortugas (islands west of Key West, Florida). Data are currently not adequate to assess trends in the annual number of nests for this subpopulation;

(4) Northern Gulf of Mexico (Franklin County, Florida, west through Texas). Data are currently not adequate to assess trends in the annual number of nests for this subpopulation; and

(5) Greater Caribbean (Mexico through French Guiana, the Bahamas, Lesser and Greater Antilles). This is the third largest subpopulation within the Northwest Atlantic population, with the majority of nesting at Quintana Roo, Mexico. TEWG (2009) reported a greater than 5 percent annual decline in loggerhead nesting from 1995–2006 at Quintana Roo, however, a reanalysis using nesting counts from a longer time series (1989 through 2010) for this subpopulation did not detect a trend (a trend not significantly different from zero as compared with the subpopulation for which there are enough data to analyze).

Diving Behavior

Studies of loggerhead diving behavior indicate varying mean depths and surface intervals, depending on whether they were located in shallow coastal areas (short surface intervals) or in deeper, offshore areas (longer surface intervals). The maximum recorded dive depth for a post-nesting female was 211-233 meters, while mean dive depths for both a post-nesting female and a subadult were 9-22 meters. Routine dive times for a post-nesting female were between 15 and 30 minutes, and for a subadult, between 19 and 30 minutes (Sakamoto *et al.* 1990 *cited in* Lutcavage and Lutz 1997). Two loggerheads tagged by

Hawaii-based longline observers in the North Pacific and attached with satellite-linked dive recorders were tracked for about 5 months. Analysis of the dive data indicates that most of the dives were very shallow - 70% of the dives were no deeper than 5 meters. In addition, the loggerheads spent approximately 40% of their time in the top meter and nearly all of their time at depths shallower than 100 meters. On 5% of the days, the turtles dove deeper than 100 meters; the deepest daily dive recorded was 178 meters (Polovina *et al.* 2003).

Polovina *et al.* (2004) reported that tagged turtles spent 40 percent of their time at the surface and 90 percent of their time at depths shallower than 40 meters. On only five percent of recorded dive days loggerheads dove to depths greater than 100 meters at least once. In the areas that the loggerheads were diving, there was a shallow thermocline at 50 meters. There were also several strong surface temperature fronts the turtles were associated with, one of 20°C at 28°N latitude and another of 17°C at 32°N latitude.

Free swimming hatchlings typically swim at the surface or within the upper 1 m of the water column with occasional diving to about 3-4 m (Wyneneken 1997).

Hearing

The information on loggerhead turtle hearing is very limited. Bartol *et al.* (1999) studied the auditory evoked potential of loggerhead sea turtles that had been captured in pound nets in tributaries to the Chesapeake Bay in Maryland and Virginia and concluded that loggerhead sea turtles had most sensitive hearing between 250 and 1000 Hz, with rapid decline above 1000 Hz (Bartol *et al.* 1999, Lenhardt 1994; Lenhardt et al. 1983; Moein Bartol and Ketten 2006; Moein Bartol et al. 1999; O'Hara and Wilcox 1990). This is similar to the results produced by Ridgway *et al.* (1969) who studied the auditory evoked potentials of three green sea turtles (in air and through mechanical stimulation of the ear). They concluded that the maximum sensitivity of green sea turtles occurred from 300 to 400 Hz with rapid declines for tones at lower and higher frequencies. They found that 2,000 Hz was the upper limit of maximum sensitivity without injury for auditory evoked potentials and suggested that the practical hearing range was no more than 1,000 Hz.

Atlantic Sturgeon

Distribution

The Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus*) is a subspecies of sturgeon distributed along the eastern coast of North America from Hamilton Inlet, Labrador, Canada to the Saint Johns River in Florida (Smith and Clugston 1997, ASSRT 2007). Atlantic sturgeon have been documented as far south as

Bermuda and Venezuela (Lee *et al.* 1980). Historically, Atlantic sturgeon were present in approximately 38 rivers in the United States from St. Croix, ME to the Saint Johns River, FL, of which 35 rivers have been confirmed to have had historic spawning populations. Atlantic sturgeon are currently present in 36 rivers, and spawning occurs in at least 20 of these. Other estuaries along the coast formed by rivers that do not support Atlantic sturgeon spawning populations may still be important rearing habitats.

Atlantic sturgeon are long lived, late maturing, estuarine dependent, anadromous fish. Atlantic sturgeon spawn in freshwater, but spend most of their sub-adult and adult life in the marine environment. The spawning areas in most U.S. rivers have not been well defined. However, the habitat characteristics of spawning areas have been identified based on historical accounts of where fisheries occurred, tracking and tagging studies of spawning sturgeon, and physiological needs of early life stages. Spawning is believed to occur in flowing water between the salt front of estuaries and the fall line of large rivers.

Larval Atlantic sturgeon (i.e. less than 4 weeks old, with total lengths (TL) less than 30 mm; Van Eenennaam et al. 1996) are assumed to undertake a demersal existence and inhabit the same riverine or estuarine areas where they were spawned (Smith et al. 1980; Bain et al. 2000; Kynard and Horgan 2002; ASMFC 2009). Studies suggest that age-0 (i.e., young-of-year), age- 1, and age-2 juvenile Atlantic sturgeon occur in low salinity waters of the natal estuary (Haley 1999; Hatin et al. 2007; McCord et al. 2007; Munro et al. 2007) while older fish are more salt tolerant and occur in higher salinity waters as well as low salinity waters (Collins et al. 2000). Atlantic sturgeon remain in the natal estuary for months to years before emigrating to open ocean as subadults (Holland and Yelverton 1973; Dovel and Berggen 1983; Waldman et al. 1996; Dadswell 2006; ASSRT 2007).

After emigration from the natal estuary, subadults and adults travel within the marine environment, where they may undergo extensive movements usually confined to shelly or gravelly bottoms in 10-50 m (33-164 ft) water depths (Stein et al., 2004; Erickson et al., 2011). Fish distribution varies seasonally within this depth range. During summer months (May to September) fish are primarily found in the shallower depths of 10-20 m (33-66 ft). In winter and early spring (December to March), fish move to depths between 20 and 50 m (66 and 165 ft) (Erickson et al., 2011). Shelf areas <18 m (59 ft) deep off Virginia and the sandy shoals offshore of Oregon Inlet, North Carolina, appear to be areas of concentration during summer months (Laney et al., 2007). The area of high concentration offshore of Virginia was centered from 15 to 37.5 km (9.3 to 23.3 mi) from shore, and the maximum distance from shore during winter was about 112.5 km (70 mi).

After leaving the Delaware River estuary during the fall, juvenile Atlantic sturgeon were recaptured by commercial fishermen in nearshore waters along the Atlantic coast as far south as Cape Hatteras, North Carolina from November through early March. In the spring, a portion of the tagged fish reentered the Delaware River estuary. However, many fish continued a northerly coastal migration through the Mid-Atlantic as well as into southern New England waters where they were recovered throughout the summer months. Movements as far north as Maine were documented. A southerly coastal migration was apparent from tag returns reported in the fall. The majority of these tag returns were reported from relatively shallow near shore fisheries with few fish reported from waters in excess of 25 m (C. Shirey, Delaware Department of Fish and Wildlife, unpublished data reviewed in ASMFC, 2009). Areas where migratory Atlantic sturgeon commonly aggregate include the Bay of Fundy (e.g., Minas and Cumberland Basins), Massachusetts Bay, Connecticut River estuary, Long Island Sound, New York Bight, Delaware Bay, Chesapeake Bay, and waters off of North Carolina from the Virginia/North Carolina border to Cape Hatteras at depths up to 24 m (Dovel and Berggren, 1983; Dadswell et al., 1984; Johnson et al., 1997; Rochard et al., 1997; Kynard et al. 2000; Eyler et al. 2004; Stein et al.2004; Wehrell 2005; Dadswell 2006; ASSRT 2007; Laney et al. 2007). These sites may be used as foraging sites and/or thermal refuge.

Population Structure

NMFS has delineated U.S. populations of Atlantic sturgeon into five DPSs (77 FR 5880 and 77 FR 5914). These are: the Gulf of Maine (GOM), New York Bight (NYB), Chesapeake Bay (CB), Carolina, and South Atlantic (SA) DPSs. The results of genetic studies suggest that natal origin influences the distribution of Atlantic sturgeon in the marine environment (Wirgin and King 2011). However, genetic data as well as tracking and tagging data demonstrate sturgeon from each DPS and Canada occur throughout the full range of the subspecies. Therefore, individuals originating from all of the 5 listed DPSs may occur in the Action Area. Information general to all Atlantic sturgeon as well as information specific to each of the relevant DPSs is provided.

Threats to the Species

Natural threats. Atlantic sturgeon occur naturally in small populations and in some rivers because of variable spawning returns, Allee affects could be an issue. During all stages of development, Atlantic sturgeon are sensitive to temperatures above 28°C (Niklitschek and Secor 2005, Kahn and Mohead 2010, Niklitschek and Secor 2010) and dissolved oxygen levels below 4.3 to 4.7 parts per million (Secor and Niklitschek 2002, EPA 2003, Niklitschek and Secor 2009). Juvenile sturgeon are also stressed by high salinities until they mature and out migrate.

Anthropogenic threats. Historical records provide evidence of commercial fisheries for Atlantic sturgeon in the Kennebec and Androscoggin Rivers dating back to the 17th century (Squiers et al. 1979); and in the James River and Chesapeake Bay in the 19th century (Hildebrand and Schroeder 1928; Vladykov and Greeley 1963; ASMFC 1998; Secor 2002; Bushnoe et al. 2005; ASSRT 2007). Following the 1880s, the sturgeon fishery was almost non-existent due to a collapse of the sturgeon stocks.

All directed Atlantic sturgeon fishing as well as retention of Atlantic sturgeon bycatch has been prohibited since 1998. Nevertheless, mortalities associated with bycatch in fisheries occurring in state and federal waters still occur throughout the marine range, as well as in some riverine waters, reducing survivorship of subadult and adult Atlantic sturgeon (Stein et al. 2004; ASMFC 2007). Stress or injury to Atlantic sturgeon taken as bycatch but released alive may result in increased susceptibility to other threats, such as poor water quality (e.g., exposure to toxins and low DO). This may result in reduced ability to perform major life functions, such as foraging and spawning, or even postcapture mortality.

The Atlantic Sturgeon Status Review Team (2007) determined Atlantic sturgeon in the Delaware River are at a moderately high risk of extinction because of ship strikes and sturgeon in the James River are at a moderate risk from ship strikes. Since that time, managers in the Hudson River are concerned that ship strikes may also be threatening Atlantic sturgeon populations there. In these systems, large ships move upstream from the mouths of the river to ports upstream through narrow shipping channels. The channels are dredged to the approximate depth of the ships, usually leaving less than 6 feet of clearance between the bottom of ships and the benthos of the river. Because of the size of the propellers used on large ships, everything along the bottom is sucked through the propellers. Large sturgeon are most often killed by ship strikes because smaller fish often pass through the propellers without making contact but larger sturgeon get hit. As shipping increases in the future, as has been predicted by the US Coast Guard, more Atlantic sturgeon are likely to be killed during encounters with ships. Besides the threats to Atlantic sturgeon from ships, the act of dredging the channel can also kill sturgeon.

Twenty-nine mortalities believed to be the result of vessel strikes were documented in the Delaware River from 2004 to 2008, and at least 13 of these fish were large adults. Given the time of year in which the fish were observed (predominantly May through July, with two in August), it is likely that many of the adults were migrating through the river to the spawning grounds. Eleven Atlantic sturgeon were reported to have been struck by vessels from 2005 through 2007 in the James River; several of these were mature individuals.

Dredging projects in the rivers such as the Kennebec, Delaware, James, Cape Fear, and Savannah Rivers put Atlantic sturgeon at moderate risk (ASSRT 2007). Dredging primarily affects sturgeon by removing food resources and homogenizing habitat, eliminating holding areas and other high quality habitat. Also, sometimes Atlantic sturgeon are attracted to the sediment plume created during dredging operations and are killed by the dredge itself. While some dredging projects operate with observers present to document fish mortalities, many do not.

Connectivity is disrupted by the presence of dams on several rivers, including the Penobscot and Merrimack Rivers, whereas other dams, such as those on the Kennebec, Androscoggin and Saco Rivers, are near the site of natural falls and likely represent the maximum upstream extent of sturgeon occurrence even if the dams were not present. Dams have curtailed Atlantic sturgeon spawning and juvenile developmental habitat by blocking over 60 percent of the historical sturgeon habitat upstream of the dams in the Cape Fear and Santee-Cooper River systems.

Most Atlantic sturgeon managers and researchers consider water quality as a moderate risk to every DPS in the United States (ASSRT 2007). Atlantic sturgeon are sensitive to pesticides, heavy metals, and other toxins in the aquatic environment.

Many rivers in Maine, including the Androscoggin River, were heavily polluted in the past from industrial discharges from pulp and paper mills. Both the Hudson and Delaware rivers, as well as other rivers in the New York Bight region, were heavily polluted in the past from industrial and sanitary sewer discharges. While water quality has improved (Lichter et al. 2006; EPA 2008) and most discharges are limited through regulations, many pollutants persist in the benthic environment. This can be particularly problematic if pollutants are present on spawning and nursery grounds as developing eggs and larvae are particularly susceptible to exposure to contaminants.

The Chesapeake Bay system is vulnerable to the effects of nutrient enrichment due to a relatively low tidal exchange and flushing rate, large surface to volume ratio, and strong stratification during the spring and summer months (Pyzik et al. 2004; ASMFC 1998; ASSRT 2007; EPA 2008). These conditions contribute to reductions in dissolved oxygen levels throughout the Bay, which in turn can limit the availability of nursery habitat (Niklitschek and Secor 2005; 2010). Similarly, low dissolved oxygen levels in the Roanoke, Savannah, Ogeechee, St. Marys, and St Johns Rivers due to dredging and non-point source inputs affect or even eliminate nursery habitat. Large water withdrawals can alter flows, temperature, and dissolved oxygen levels, In the Pamlico and Neuse systems, nutrient loading and seasonal anoxia are occurring, associated in part with concentrated animal feeding operations (CAFOs). Heavy industrial development and CAFOs have degraded water quality in the Cape Fear River. Water quality in the Waccamaw and Pee Dee rivers have been affected by industrialization and riverine sediment samples contain high levels of various toxins, including dioxins. Additional stressors arising from water allocation and climate change threaten to exacerbate water quality problems that are already present throughout the range of the Carolina DPS. Twenty interbasin water transfers in existence prior to 1993, averaging 66.5 million gallons per day (mgd), were authorized at their maximum levels without being subjected to an evaluation for certification by North Carolina Department of Environmental and Natural Resources or other resource agencies. Since the 1993 legislation requiring certificates for transfers, almost 170 mgd of interbasin water withdrawals have been authorized, with an additional 60 mgd pending certification. The removal of large amounts of water from the system will alter flows, temperature, and DO. Existing water allocation issues will likely be compounded by population growth and potentially climate change.

Status

Prior to 1890, Atlantic sturgeon populations were at or near carrying capacity. In the mid-1800s, incidental catches of Atlantic sturgeon in the shad and river herring haul seine fisheries indicated that the species was very abundant (reviewed in Armstrong and Hightower 2002). A major fishery for this species did not exist until 1870 when a caviar market was established (reviewed in Smith and Clugston 1997). Record landings were reported in 1890, where over 3350 metric tons (mt) of Atlantic sturgeon were landed from coastal rivers along the Atlantic Coast (reviewed in Smith and Clugston 1997, Secor and Waldman 1999). Between 1890 and 1905, Atlantic sturgeon (and shortnose sturgeon) populations were drastically reduced for sale of meat and caviar. Between 1920 and 1998, the harvest level remained very low due to small remnant populations. The majority of these landings (75%) were dominated by the Delaware River fishery, which presumably supported the largest population along the Atlantic Coast (reviewed in Secor and Waldman 1999). Prompted by research on juvenile production between 1985 and 1995 (Peterson *et al.* 2000), the Atlantic sturgeon fishery was closed by the Atlantic States Marine Fisheries Commission in 1998, when a coastwide fishing moratorium was imposed for 20 to 40 years, or at least until 20 year classes of mature female Atlantic sturgeon were present (ASMFC 1998).

Ten years after peak landings, the fishery collapsed in 1901, when less than 10% (295 mt) of its 1890 peak landings were reported. The landings continued to decline to about 5% of the peak until 1920 and have remained between 1-5% since then. During the 1950s, the remaining fishery switched to targeting sturgeon for flesh, rather than caviar. The Atlantic sturgeon fishery was closed by the Atlantic States

Marine Fisheries Commission in 1998, when a coastwide fishing moratorium was imposed for 20-40 years, or at least until 20 year classes of mature female Atlantic sturgeon were present (ASMFC 1998).

There are no current, published population abundance estimates for any of the currently known spawning stocks. Therefore, there are no published abundance estimates for any of the five DPSs of Atlantic sturgeon. An estimate of 863 mature adults per year (596 males and 267 females) was calculated for the Hudson River based on fishery-dependent data collected from 1985-1995 (Kahnle et al. 2007). Peterson *et al.* (2008) reported that approximately 324 and 386 adults per year returned to the Altamaha River in 2004 and 2005, respectively. Using the data collected from the Hudson River and Altamaha River to estimate the total number of Atlantic sturgeon in either subpopulation is not possible, since mature Atlantic sturgeon may not spawn every year (Vladykov and Greeley 1963; Smith 1985; Van Eenennaam et al. 1996; Stevenson and Secor 1999; Collins et al. 2000; Caron et al. 2002), the age structure of these populations is not well understood, and stage to stage survival is unknown.

In other words, the information that would allow us to take an estimate of annual spawning adults and expand that estimate to an estimate of the total number of individuals (e.g., yearlings, subadults, and adults) in a population is lacking. The Atlantic Sturgeon Status Review Team (ASSRT) presumed that the Hudson and Altamaha rivers had the most robust of the remaining U.S. Atlantic sturgeon spawning populations and concluded that the other U.S. spawning populations were likely less than 300 spawning adults per year (ASSRT 2007).

Juvenile Atlantic sturgeon abundance may be a more precise way to measure the status of Atlantic sturgeon populations because it is believed that all age-1 and age-2 juveniles are restricted to their natal rivers (Dovel and Berggren 1983, Bain *et al.* 1999), avoiding the assumptions noted above. Peterson *et al.* (2000) reported that there were approximately 4,300 age-1 and -2 Atlantic sturgeon in the Hudson River between 1985 and 1995. Schueller and Peterson (2010) reported that age-1 and -2 Atlantic sturgeon population densities ranged from 1,000 to 2,000 individuals over a 4 year period from 2004 to 2007. Other spawning populations within the U.S. are predicted to have fewer than 300 adults spawning per year.

Hearing

Information available about the hearing abilities of Atlantic sturgeon come from studies of other species of sturgeon.

Meyer and Popper (2003), investigated shortnose sturgeon *(Acipenser brevirostrum*) hearing abilities by using physiological methods to measure responses to pure tones. The authors presented shortnose sturgeon with pure tone stimuli from 50-1000 Hz with intensities ranging from of 120 to 160 dB re 1 µpa. Shortnose sturgeon were most sensitive to tones presented at 100 and 400 Hz although thresholds were not determined. Based on the limited data, sturgeon were able to detect sounds below 100 Hz to about 1,000 Hz and that sturgeon should be able to determine the direction of sounds (Popper 2005).

Meyer and Popper (2005) recorded auditory evoked potentials to pure tone stimuli of varying frequency and intensity in lake sturgeon and reported that lake sturgeon detect pure tones from 100 to 2000 Hz, with best sensitivity from 100 to 400 Hz. They also compared these sturgeon data with comparable data for oscar (*Astronotus ocellatus*) and goldfish (*Carassius auratus*) and reported that the auditory brainstem responses for the lake sturgeon are more similar to the goldfish (which is can hear up to 5000 Hz) than to the oscar (which can only detect sound up to 400 Hz); these authors, however, felt additional data were necessary before lake sturgeon could be considered in the same auditory group as the goldfish.

Lovell et al. (2005) also studied sound reception in and the hearing abilities of paddlefish (*Polyodon spathula*) and lake sturgeon (*Acipenser fulvescens*) in pressure dominated and particle motion dominated sound fields. They concluded that both species were responsive to sounds ranging in frequency from 100 to 500 Hz with lowest hearing thresholds from frequencies in bandwidths between 200 and 300 Hz and higher thresholds at 100 and 500 Hz. The results showed that both species were not sensitive to sound pressure, and would have a significantly higher hearing threshold in a pressure dominated sound field. Based on the above we assume that the hearing sensitivity of shortnose sturgeon is best between 100-500 Hz with sensitivity falling up to 1,000 Hz.

BOEM (2012c) categorized sturgeon in general as fishes that detect sounds from below 50 Hz to perhaps 800-1,000 Hz (though several probably only detect sounds to 600-800 Hz). These fishes have a swim bladder but no known structures in the auditory system that would enhance hearing, and sensitivity (lowest sound detectable at any frequency) is not very great. Sounds would have to be more intense to be detected compared to fishes with swim bladders that enhance hearing. Sturgeon can detect both particle motion and pressure.

Environmental Baseline

By regulation, environmental baselines for biological opinions include the past and present impacts of all state, Federal or private actions and other human activities in the Action Area, the anticipated impacts of all proposed Federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of State or private actions which are contemporaneous with the consultation in process (50 CFR 402.02). The environmental baseline for this biological opinion includes the effects of several activities that affect the survival and recovery of endangered whales, threatened and endangered sea turtles and Atlantic sturgeon in theAction Area.

A number of human activities have contributed to the current status of populations of large whales, sea turtles and anadromous fish in the action area. Some of those activities, most notably commercial whaling, occurred extensively in the past, continue at low levels, and no longer appear to affect these whale populations, although the effects of these reductions likely persist today. Other human activities are ongoing and appear to continue to affect whale and sea turtles populations in the Action Areas for this consultation. The following discussion summarizes the principal phenomena that are known to affect these whales, sea turtles and Atlantic sturgeon in the Action Area.

Natural Mortality

The sources of natural mortality discussed in the species-specific narratives from the preceding section of this Opinion affect endangered and threatened whales, sea turtles and Atlantic sturgeon in the Action Areas for this consultation as well. Large whale species are known to be affected by parasites. The occurrence of the nematode *Crassicauda boopis* appears to increase the potential for kidney failure in large whales and may be preventing some fin and humpback whale populations from recovering (Lambertsen, 1992). Killer whale and shark attacks have also been documented in blue, fin, sei and humpback whales and may result in serious injury or death in very young and sick individuals (Perry, DeMaster, & Silber, 1999).

The various habitat types sea turtles occupy along the Atlantic coast of the United States exposes these sea turtles to a wide variety of natural threats. Hatchling sea turtles are hunted by predators like herons, gulls, dogfish, and sharks. Adult sea turtles are also killed by sharks and other large, marine predators and are killed by cold stunning and exposure to biotoxins. The Atlantic beaches on which loggerhead, green and leatherback sea turtles nest and the nests themselves are threatened by hurricanes and tropical storms as well as the storm surges, sand accretion and rainfall that are associated with these types of storms. The nesting ecology of sea turtle species determines their vulnerability to major storm systems. For example, from 1995 to 2005, Pike and Stiner (2007) monitored nesting beaches along an uninhabited portion of Florida’s Atlantic coast. They compared the number of successful reproductive events (number of nest hatched and the number of hatchlings that emerged) with the number of tropical storms and hurricanes as well as storm induced flooding. These authors report that since leatherback turtles began nesting earliest and most eggs hatched before the tropical cyclone season arrived, they were not affected by storm surges caused by major storm systems. The loggerhead turtle nesting season began before and continued into the storm season causing a portion of nests to be inundated by storm surges while the green turtle nesting season occurred during the entire tropical cyclone season leaving 79% of green turtle nests incubating in September susceptible to storm surge.

For Atlantic sturgeon, bony scutes likely protect these fish from predation once they reach a total length of 25 mm (Gadomski and Parsley 2005 *In* ASSRT 2007. Trematodes as well as nematodes have been found infesting juvenile and adult sturgeon, respectively. Parasites are not uncommon in fish species, however, diseases from parasite do not normally occur unless some environmental event alters the equilibrium between the parasites and the fish (ASSRT 2007). During all stages of development, Atlantic sturgeon are sensitive to temperatures above 28°C (Niklitschek and Secor 2005, Kahn and Mohead 2010, Niklitschek and Secor 2010) and dissolved oxygen levels below 4.3 to 4.7 parts per million (Secor and Niklitschek 2002, EPA 2003, Niklitschek and Secor 2009). Juvenile sturgeon are also stressed by high salinities until they mature and out migrate.

Human-Induced Mortality

Commercial Whaling and Subsistence Hunting

Large whale population numbers in the Action Area have historically been impacted by commercial exploitation, mainly in the form of whaling. Prior to current prohibitions on whaling, such as the International Whaling Commission's 1982 moratorium on commercial whaling on all whale stocks, most large whale species had been depleted to the extent it was necessary to list them as endangered under the ESA of 1966. Nevertheless, fin and humpback whales are still hunted in subsistence fisheries in the northwest Atlantic based on IWC aboriginal subsistence catch limits and by Iceland under objection to the IWC moratorium. Although these subsistence fisheries operate outside of the Action Area, the fin whales killed in these fisheries are part of the population of fin whales occurring within the Action Area for this consultation. From 1985 to 2011 a total of 317 fin whales have been taken in the West Greenland and St Vincent and The Grenadines subsistence fisheries (IWC 2012l). These totals include adult and juvenile whales of both sexes that have been killed and landed as well as struck and lost. Based on abundance estimates (4,539 95% CI 1,897-10,114 for fin whales; the IWC believes that this level of removal will not reduce the growth rates of these populations (IWC2012m). These fisheries are expected to continue into the future as Greenland has requested 19 fin whales for the 2013-2018 whaling seasons (IWC 2012n). Iceland, however, continues its commercial whaling program (suspended since 1986) with 7 fin whales caught in 2006, 125 in 2009 and 148 (74 males and 68 females with 6 struck and lost) in 2010 (IWC 2012o).

Ship Traffic

The North Atlantic is one of the most traveled areas in the world for marine shipping. Between 2002 and 2010 port calls between Delaware and Port Canaveral increased from a low of 21,257 in 2003 to 27, 038 in 2010 for a total of 99,757 vessel arrivals over the 8-year period (See Table 8; U.S. Department of Transportation 2011). Virginia ports received 3, 021 vessel calls from all type of vessels, surpassed only by Houston, Texas (6,698 arrivals), New York/New Jersey (4,534 arrivals) and Long Beach/Los Angeles, California (4,469 arrivals) in 2010 (U.S. Department of Transportation 2011). Other eastern seaboard ports such as Baltimore, Maryland (2,011 arrivals) Savannah, Georgia (2,406), Charleston, South Carolina (1,818 arrivals) and Jacksonville, Florida (1,641 arrivals) also receive significant numbers of arrivals.

Expansion of the Panama Canal is scheduled for completion in 2014 and commercial maritime traffic off the U.S. Atlantic and Gulf coasts is expected to increase as cargo traffic shifts to take advantage of shorter routes. U.S. port container traffic is expected to double or triple by 2030 (DOT 2009). Shipping companies are also maximizing each vessel call with larger capacity containerships. Between 2005 and 2010 companies are increasing their use of Post-Panamax class containerships increasing the average size of ships by 7 percent. At 1,200 ft long, 160 ft beam (wide) and 50 ft draft, these containerships are about two and a half times the capacity of vessels that currently are small enough to transit the canal (965 ft long, 106 ft beam and 39.5 ft draft) (U.S. Department of Transportation 2010). Increased dredging to deepen and widen port channel entrances and improved on-shore infrastructure such as cranes, railways and roadways to distribute arriving cargo are necessary to accommodate these deep-draft vessels (panama class vessels in U.S. ports (Chambers 2010). To date, only the port in Savannah, Georgia has the capacity to accommodate this new class of cargo ships

Table 8: 2002-2010 Vessel Calls at Ports Located Between Delaware and Cape Canaveral, Florida.

Year	Calls of All Types	Tanker Calls	Product Tanker Calls	Crude Tanker Calls	Container Calls	Dry Bulk Calls	Ro-Ro Calls	Vehicle Calls	Gas Carrier Calls	Combi nation Calls	General Cargo Calls
2002	10,094	1,084	1,052	32	4,793	1,419	1,899	1,327	24	75	800
2003	9,466	1,121	1,085	36	4,428	1,287	1,810	1,204	69	64	687
2004	10,827	1,221	1,166	55	5,146	1,629	1,840	1,166	146	88	757
2005	11,233	1,383	1,301	82	5301	1,608	1,998	1534	145	85	639
2006	11,969	1,429	1,343	86	5,788	1,790	2,147	1,649	104	75	636
2007	11,952	1443	1,349	94	6137	1,469	2,149	1,657	83	75	596
2008	11,735	1,419	1,281	138	5693	1,680	2,294	1,774	87	46	516
2009	10,536	1,147	998	149	5,646	1,169	2,007	1,525	84	15	468
2010	11,945	1,231	1,119	112	6,043	1,564	2,471	1,917	56	45	535
2002-2010 TOTALS											
	99,757	11,478	10,694	784	48,975	13,615	18,606	13,753	718	568	5,634

About 302 Federal vessels have been estimated to operate in waters off the East Coast, although all of these vessels probably do not operate at one time or in the same area (NOAA 2008). The percentage of time these vessels spend at sea varies with the specific mission and objectives of each agency. For example, a study conducted on Navy vessel traffic estimated that of the Navy's 121 East Coast vessels, there are 12 vessels on the East Coast within 200 nm (370.4 km) of shore at any given time (Filadelfo 2001).

Ship Strikes and Entanglements

As discussed in the *Status of the Species* narratives for several of the whales that are considered in this Opinion, ship strikes and entanglement in fishing gear pose significant threats to populations of endangered whales along the Atlantic seaboard, particularly North Atlantic right whales. Based on the records available, large whales have been struck by ships off almost every coastal state in the United States, although ship strikes are most common along the Atlantic Coast. More than half (56 percent) of the recorded ship strikes from 1975 to 2002 occurred off the coasts of the northeastern United States and Canada, while the mid- Atlantic and southeastern areas each accounted for 22 percent (Jensen and Silber 2003).

In particular, ship strikes represent the greatest threat to the continued existence of North Atlantic right whales: between 1999 and 2006, ships are confirmed to have struck 22 North Atlantic right whales, killing 13 of these whales (Jensen and Silber 2003, Knowlton and Kraus 2001, NMFS 2005b).

Glass *et al*. (2008) reported that there were 54 instances in which right whales were entangled, struck by vessels or otherwise stranded between 2002 and 2006. Out of 21 verified right whale mortalities, 10 were from ship strikes and 3 were from entanglement. Entanglement was identified as the cause of four recorded serious injuries. There were also two documented serious injuries from ship strikes (Glass *et al*. 2008). From 2006 through 2010, there were 55 interactions involving right whales (Henry et al 2012). Of the nineteen confirmed mortalities and injuries serious enough to cause death, four were attributed to entanglements and 5 to vessel strikes (Henry et al 2012).

From 2006 to 2010, Henry et al (2012) reported a total of 500 large whale interactions including ship strikes, entanglements and strandings due to unknown causes. Two hundred-sixty four (53%) interactions involved humans, 16 (3%) did not involve humans and for 220 (44%) of the events it was unknown if humans were involved. The number of entanglements (206) far outweighed the number of vessel strikes (58); however, 4 events had evidence of both entanglement and vessel strike. Among the entanglements 24 (12%) caused the deaths of the individuals and 28 (48%) of the vessel strikes caused the deaths of the animals. Thirty-three (16%) of the entanglements were serious enough to result in death of the animal while only1 of the vessel strikes resulted in a serious injury. Disentanglement teams were successful in preventing serious injuries in 28 (14%) entanglement events. Although 45 (22%) of entanglements and 9 (16%) of the vessel strike events did not have adequate documentation to determine if serious injury occurred, 73 (35%) of the entanglement events and 17 (29%) of the vessel strike events were determined to have not caused serious injury or death. Out of a total of 313 confirmed mortalities, 52 (17%) were attributed to human interactions, 16 (5%) were attributed to natural causes and 245 (78%) lacked sufficient evidence to determine the cause of death. Humpback whales had the greatest number of entanglement mortalities (n=9), the highest number of serious injury events resulting from entanglements (n=20); and the greatest number of vessel strike mortalities (n=10); and right whales had the only serious injury (n=1) from vessel strikes.

A summary paper on ship collisions and whales by Laist *et al.* (2001) reported that out of 28 recorded collisions resulting in lethal or severe injuries to whales in which vessel speed was known, 89 percent involved vessels traveling at 14 knots or faster and the remaining 11 percent involved vessels traveling at 10 to 14 knots. None occurred at speeds below 10 knots. The IWC database of vessel collisions identified 83 events where speed was recorded; the majority of serious injuries and mortalities occurred within a similar range of 15 to 20 knots (Van Waerbeek and Leaper 2008). With regard to the severity of injuries at increasing speeds, Pace and Silber (2005) found a predicted 45 percent chance of death or serious injury at 10 knots. Vanderlaan and Taggart (2007) came to a similar conclusion, determining that the probability of death from a collision was approximately 35-40 percent at 10 knots while Wiley et al

(2011) predicted a 29 percent chance of death at 10 knots. Furthermore, Silber et al (2010) reported that as vessel speeds increased the hydrodynamic pull of vessels on large whales within proximity of those vessels could make whales more vulnerable to strikes.

North Atlantic right whales appear to be either unable to detect approaching vessels or, while right whales are engaged in behavioral activities — for example, feeding, nursing, or mating — they ignore the visual or acoustic cues those vessels produce. Because right whales are buoyant and are slow swimmers, they may not be able to avoid oncoming vessels even if they are aware of its approach. When the vulnerability of right whales to ship strikes is combined with the density of ship traffic within the distribution of right whales, ship strikes seem almost inevitable.

Various types and sizes of vessels have been involved in ship strikes with large whales, including container/cargo ships/freighters, tankers, steamships, U.S. Coast Guard vessels, Navy vessels, cruise ships, ferries, recreational vessels, fishing vessels, whale-watching vessels, and other vessels (Jensen and Silber 2003). Vessel speed (if recorded) at the time of a large whale collision has ranged from 2 to 51 knots (Jensen and Silber, 2003). Vessels can be damaged during ship strikes (occasionally, collisions with large whales have even harmed or killed humans on board the vessels); of 13 recorded vessels that reported damages from a strike, all were traveling at a speed of at least 10 knots (Jensen and Silber 2003).

Atlantic sturgeon are also susceptible to vessel collisions. Out of a total of 28 mortalities reported in the Delaware estuary between 2005 and 2008, 14 resulted from vessel strike (Brown and Murphy 2007). Based on the demersal behavior demonstrated by Atlantic sturgeon, the damage inflicted upon carcasses and the large numbers of deep draft vessels, the authors concluded that interactions with large vessels such as tankers comprised the majority of the vessel strikes. Further, the authors determined that a mortality rate of more than 2.5% of the females within a population could result in population declines. Similarly, in the James River in Virginia, 34 out of a total of 39 Atlantic sturgeon had injuries consistent with vessel strikes (Brown and Murphy 2007, Balazik et al 2012). The actual number of vessel strikes in both of these river systems in unknown, however, Balazik et al (2012) estimated up to 80 sturgeon were killed between 2007 and 2010.

For sea turtles, vessels also present risks for injury and death. For example, out of a total of 109 stranded, dead sea turtles found in the Delaware River estuary from 1994-1999, 30 had been struck and killed by vessels (Stetzar 2002). Numbers of vessel struck sea turtles would be elevated above those mentioned here in the southern part of their ranges where sea turtle abundance is greater.

Several commercial fisheries operate in the Action Area for this consultation. The shrimp trawl fisheries conducted off the southeast United States (from North Carolina to the Atlantic coast of Florida) and Gulf of Mexico (from the Gulf coast of Florida to Texas) are one of the fisheries that have the most significant demographic effect on sea turtles. Participants in these fisheries are required to use Turtle Exclusion Devices (TEDs), which are estimated to reduce the number of shrimp trawl related mortality by as much as 94% for loggerheads and 97% for leatherbacks. Even with TED measures in place, in 2002, NMFS (2002) expected these fisheries to capture about 323,600 sea turtles each year and kill about 5,600 of the turtles captured. Loggerhead sea turtles account for most of this total: 163,000 captured, killing almost 4,000 of them. Kemp's ridleys account for the second-most interactions: 155,503 captures with 4,200 of them dying. These are followed by green sea turtles: about 18,700 captured with more than 500 of them dying as a result of their capture. Leatherback sea turtle interactions were estimated at 3,090 captures with 80 of them dying as a result (NMFS 2002). Since 2002, however, effort in the Atlantic shrimp fisheries has declined from a high of 25,320 trips in 2002 to approximately 13,464 trips in 2009. Since sea turtle takes are directly linked to fishery effort, these takes are expected to decrease but are still expected to result in at least hundreds and possibly in the low thousands of sea turtle interactions annually, of which hundreds are expected to be lethal (NMFS 2012).

Portions of the Atlantic pelagic fisheries for swordfish, tuna, shark, and billfish also operate in the Action Area and capture and kill the second highest numbers of sea turtles along the Atlantic coast. These fisheries, which operate off the coast of South Carolina and Georgia (with the exception of waters off Florida and southernmost Georgia that are closed to the longline component of these fisheries) and the Gulf of Mexico, include purse seine fisheries for tuna, harpoon fisheries for tuna and swordfish, commercial and recreational rod and reel fisheries, gillnet fisheries for shark, driftnet fisheries, pelagic longline fisheries, and bottom longline fisheries.

Between 1986 and 1995, this fishery captured and killed 1 northern right whale, 2 humpback whales, and two sperm whales. Between 1992 and 1998, the longline components of these fisheries are estimated to have captured more than 10,000 sea turtles (4,585 leatherback sea turtles and 5,280 loggerhead sea turtles), killing 168 of these sea turtles in the process (the latter estimate does not include sea turtles that might have died after being released; Johnson *et al*. 1999, Yeung 1999). Since then, all components of these fisheries are estimated to capture about 2,381 sea turtles each year, killing 781 sea turtles in the process.

Portions of the Atlantic sea scallop fisheries also operate in the Action Area (off North Carolina) and capture and kill the third highest numbers of sea turtles along the Atlantic coast. These fisheries are expected to capture about 129 loggerhead sea turtles in 2012 but 49 loggerheads are estimated captured and killed in 2013 and beyond. Although these fisheries are only expected to capture 2 green, leatherback, and Kemp's ridley sea turtles each year, all of these turtles might die as a result of their capture.

In addition, sea turtles are captured and killed in several other Federal fisheries that operate along the Atlantic coast, although most of these fisheries capture and kill fewer sea turtles than the fisheries discussed in the preceding narratives. Of all the factors that influenced NMFS' decision to list sea turtles as threatened or endangered, the most significant sources of injury or mortality of juvenile, subadult, and adult sea turtles are those associated with commercial fishing. The fisheries discussed in this section of this Opinion are expected to continue into the foreseeable future at levels of effort that are roughly equivalent to current levels. As a result, we expect the number of sea turtles that are captured and killed in these fisheries to continue for the foreseeable future.

Climate Variability

Global climate patterns have a natural year-to-year variability; however, a global warming trend has become apparent based on observations of average air and sea surface temperatures, losses of ice and snow and rising average sea levels (IPCC 2001, 2007, U.S. Global Change Research Program 2009). There is now widespread consensus within the scientific community that atmospheric temperatures on earth are increasing (warming) (IPCC 2001, Oreskes 2004). Observational evidence from all continents and most oceans shows that many regional ecosystems are also being affected due to factors driving global climate change, particularly temperature increases (EPA 2010, IPCC 2007, U.S. Global Change Research Program 2009, Walther 2010). The severity of climate change depends not only on the magnitude of the change but also on the potential for irreversibility (Solomon et al 2009). Modeling on the longevity of climate-driven changes (atmospheric warming) suggest that atmospheric temperatures would not drop significantly for at least 1,000 years (even if factors driving the warming climate ceased (Solomon et al 2009.)

Annual average atmospheric temperatures in the Northeastern U.S. have been increasing by 2^{o}F since 1970 with winter temperatures rising by as much as 4^{o}F (Global Change Research Program 2009). These trends are expected to continue. Environmental changes in response to warming atmospheric temperatures already taking place include increased frequencies of extreme events (i.e., air temperatures above 90^{o}F and heavy precipitation), reduced snow and ice leading to smaller snowpacks, earlier breakup of winter ice on lakes and rivers, earlier spring freshets and peak river flows as well as rising sea levels and oceanic temperatures. In the Southeastern U.S. annual average temperatures since 1970 have risen 2^{o}F with winter temperatures experiencing the greatest increases. Hurricanes with greater intensity (greater wind speeds, rainfall and storm surge height and strength) are more likely as projections show more northern trajectories towards the mid-Atlantic. With warming temperatures water and oxygen availability in lakes, rivers and shallow habitats decrease, particularly in the already warm, humid climate of the southeast.

Atmospheric warming has direct consequences for the oceans as well. Approximately 80 percent of the rise in atmospheric temperatures is likely to be absorbed by the oceans (IPCC 2007). A recent study by Polyakov et al (2010), suggests that the North Atlantic Ocean has been experiencing a general warming trend over the last 80 years of 0.031 ± 0.006^{o}C per decade in the upper 2,000 m of the ocean. Sea surface temperatures, in the North Atlantic Ocean are closely related to the North Atlantic Oscillation. The North Atlantic Oscillation, which results from variability in pressure differences between a low pressure system that lies over Iceland and a high pressure system that lies over the Azore Islands, has been directly linked to climate changes in the North Atlantic Oceans as well as all other oceans. As these pressure systems shift from east to west, they control the strength of westerly winds and storm tracks across the North Atlantic Ocean. The North Atlantic Oscillation Index, which is positive when both systems are strong (producing increased differences in pressure that produce more and stronger winter storms) and negative when both systems are weak (producing decreased differences in pressure resulting in fewer and weaker winter storms), varies from year to year, but also exhibits a tendency to remain in one phase for intervals lasting several years. This variability results in favorable and unfavorable ecological conditions for marine species.

Changes in the oscillation and sea surface temperatures in the North Atlantic Ocean influences the abundance of marine mammal prey such as zooplankton and fish. In the 1970s and 1980s, the North Atlantic Oscillation Index has been positive and sea surface temperatures increased. These increases are believed to have produced conditions that were favorable for the copepod (*Calanus finmarchicus*), which is the principal prey of North Atlantic right whales (Conversi, Piontkovski, & Hameed., 2001) and may have increased calving rates of these whales (we cannot verify this association because systematic data on North Atlantic right whale was not collected until 1982) (C. Greene, Pershing, Kenney, & Jossi, 2003). In the late 1980s and 1990s, the North Atlantic Oscillation Index was mainly positive but exhibited two substantial, multi-year reversals to negative values. This was followed by two major, multi-year declines in copepod prey abundance (Drinkwater et al., 2003; Pershing, Head, Greene, & Jossi, 2010). Calving rates for North Atlantic right whales followed the declining trend in copepod abundance, although there was a time lag between the two (C. H. Greene et al., 2003). Copepod abundance then declined again in

1998 with a possible recovery in 1999 and 2000 followed by increased calving rates between 2001 and 2005 (Hamilton et al 2010).

Although the North Atlantic Oscillation Index has been positive for the past 25 years, atmospheric models suggest that increases in ocean temperature associated with climate change forecasts may produce more severe fluctuations in the North Atlantic Oscillation. Such fluctuations would be expected to cause dramatic shifts in the reproductive rate of critically endangered North Atlantic right whales (Drinkwater, et al., 2003; C. H. Greene, et al., 2003) and possibly a northward shift in the location of right whale calving areas (Kenney, 2007).

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. A study of relationships between climate and sea-temperature changes and the arrival of squid off southwestern England over a 20-year period concluded that veined squid (*Loligo forbesi*) migrate eastwards in the English Channel earlier when water in the preceding months is warmer, and that higher temperatures and early arrival correspond with warm phases of the North Atlantic oscillation (Sims, Genner, Southward, & Hawkins, 2001). The timing of squid peak abundance advanced by 120 - 150 days in the warmest years compared with the coldest. Ocean bottom temperatures were closely linked to the extent of squid movement and temperature increases over the five months prior to and during the month of peak squid abundance 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 is in turn mediated by climatic changes associated with the North Atlantic Oscillation. More recently, squid productivity has also been linked to sea surface temperatures in the South Atlantic, the North Pacific and the Sea of Japan indicating a global sensitivity across squid species to temperature changes with potential negative consequences if temperatures continue to rise (Postuma and Gasalla 2010).

Sperm whales, whose diets can be dominated by cephalopods, would have to re-distribute following changes in the distribution and abundance of their prey. This statement assumes that projected changes in global climate would only affect the distribution of cephalopod populations, but would not reduce the number or density of cephalopod populations. If, however, cephalopod populations collapse or decline dramatically, sperm whale populations are likely to collapse or decline dramatically as well.

Changes in global climatic patterns are also projected to have profound effect on the coastlines of every continent by increasing sea levels and increasing the intensity, if not the frequency, of hurricanes and tropical storms. 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 that are destroyed by tropical storms and hurricanes. The combination of increasing sea levels, changes in patterns of coastal erosion and accretion, and changes in rainfall patterns are likely to affect coastal estuaries, submerged aquatic vegetation, and reef ecosystems that provide foraging and rearing habitat for several species of sea turtles. Changes in ocean currents associated with climate change projections would affect the migratory patterns of sea turtles. The loss of nesting beaches, by itself, would have catastrophic effect on sea turtles populations globally if they are unable to colonize any new beaches that form of if the beaches that form do not provide the sand depths, grain patterns, elevations above high tides, or temperature regimes necessary to allow turtle eggs to survive. When combined with changes in coastal habitats and ocean currents, the future climates that are forecast place sea turtles at substantially greater risk of extinction than they already face.

Increasing sea levels, changes in patterns of coastal erosion and accretion, and changes in rainfall patterns are likely to affect coastal estuaries and submerged aquatic vegetation, that provide foraging and rearing habitat for anadromous fish. Decline in dissolved oxygen in river, stream and shallow aquatic habitats may also lead to fish kills and loss of aquatic species diversity. Decreased water availability due to increased temperatures and longer periods of time between rainfall events will ultimately affect eggs and juvenile survival resulting in changes in recruitment, abundance, distribution and growth (Drinkwater et al 2003) as well as changes in the ecosystem functions relating to the timing and availability of prey species (Walther 2010). These climate-driven hydrological changes will combine with other pressures on water resources, such as population growth and land-use change especially in coastal areas (Kundzewicz *et al*, 2008), and may place Atlantic sturgeon at a greater risk of extinction.

Habitat Degradation

Chronic exposure to the neurotoxins associated with paralytic shellfish poisoning (PSP) from zooplankton prey has been shown to have detrimental effects on marine mammals. Estimated ingestion rates are sufficiently high to suggest that the PSP toxins are affecting marine mammals, possibly resulting in lower respiratory function, changes in feeding behavior and a lower reproductive fitness (Durbin *et al.* 2002). Other human activities, including discharges from wastewater systems, dredging, ocean dumping and disposal, aquaculture and additional impacts from coastal development are also known to impact marine mammals and their habitat.

Water Pollution

Coastal runoff and river discharges carry large volumes of petrochemical and other contaminants from agricultural activities, cities and industries. Many contaminants can accumulate in marine vertebrates due to their persistence in the environment. Although these contaminant concentrations do not likely affect the more pelagic waters of the action area, the species of turtles analyzed in this biological opinion travel between nearshore and offshore habitats and may be exposed to and accumulate these contaminants during their life cycles. The contaminants that pose potential risks to the health of cetaceans, particularly North Atlantic right whales (O' Shea *et al*. 1994; Reijnders *et al*. 1999, Montie et al 2010), include persistent organic pollutants (Polychlorinated biphenyls, polychlorinated dibenzo-p-dioxins, polychlorinated dibenzofurans, dichloro-diphenyl-trichloroethane, chlordanes, and hexachlorocyclohexane); flame retardants (Polybrominated diphenyl ethers); plasticizers (Phthalate esters); surfactants (Alkyphenol ethoxylates such as nonylphenoletoxylates); new-era pesticides and herbicides; municipal and industrial effluents (including endocrine-disrupting compounds such as synthetic estrogens, natural hormones, pulp byproducts); anti-fouling agents (organotins and replacement compounds); dielectric fluids: PCB replacements (e.g., polychlorinated napthalenes, polybrominated biphenyls); aquaculture-related chemicals (such as antibiotics and pesticides); and metals such methyl mercury (Montie et al 2010).

Coastal runoff and river discharges carrying large volumes of contaminants are also detrimental to Atlantic sturgeon. Both the Hudson and Delaware rivers, as well as other rivers in the New York Bight region, were heavily polluted in the past from industrial and sanitary sewer discharges. While water quality has improved (Lichter et al. 2006; EPA 2008) and most discharges are limited through regulations, many pollutants persist in the benthic environment. This can be particularly problematic if pollutants are present on spawning and nursery grounds as developing eggs and larvae are particularly susceptible to exposure to contaminants.

Dredging

Navigation channels along the U.S. east coast are dredged and maintained to support commerce through commercial shipping and other maritime activities. Dredging primarily affects sturgeon by removing food resources and homogenizing habitat, eliminating holding areas and other high quality habitat. Also, sometimes Atlantic sturgeon are attracted to the sediment plume created during dredging operations and are killed by the dredge itself. Dickerson (2005) summarized observed takings of 24 sturgeon from dredging activities conducted by the Army Corps of Engineers and observed between 1990 and 2005 (2

Gulf; 11 shortnose; and 11 Atlantic). Hydraulic dredges (e.g., hopper) can lethally harm sturgeon and sea turtles directly by entraining them in dredge drag arms and impeller pumps. While some dredging projects operate with observers present to document mortalities, many do not. To reduce take of listed species, a boat equipped with nets precedes the dredge to capture sturgeon and sea turtles and then releases the animals out of the dredge pathway and avoiding lethal take is now standard practice.

Anthropogenic Noise

The marine mammals that occur in the action area are regularly exposed to several sources of natural and anthropogenic sounds. Anthropogenic noises that could affect ambient noise arise from the following general types of activities in and near the sea, any combination of which can contribute to the total noise at any one place and time. These noises include maritime activities, dredging, construction; mineral exploration in offshore areas; geophysical (seismic) surveys; sonars; explosions; and ocean research activities.

Noise in the marine environment has received a lot of attention in recent years and is likely to continue to receive attention in the foreseeable future. Several investigators have argued that anthropogenic sources of noise have increased ambient noise levels in the ocean over the last 50 years (Jasny *et al.* 2005; NRC, 2003, 2005; Richardson *et al.* 1995). Much of this increase is due to increased shipping as ships become more numerous and of larger tonnage (NRC 2003, Hildebrand 2009, McKenna et al. 2012). Commercial fishing vessels, cruise ships, transport boats, airplanes, helicopters and recreational boats all contribute sound into the ocean (NRC 2003; Richardson *et al.* 1995). The military uses sound to test the construction of new vessels as well as for naval operations.

Many researchers have described behavioral responses of marine mammals to the sounds produced by helicopters and fixed-wing aircraft, boats and ships, as well as dredging and construction. (Richardson *et al.* 1995). Most observations have been limited to short term behavioral responses, which included cessation of feeding, resting, or social interactions, however, habitat abandonment can lead to more long-term effects which may have implications at the population-level. Because responses to anthropogenic noise vary between species and individuals within species, it is difficult to determine long-term effects. Carretta *et al.* (2001), Jasny *et al.* (2005) and more recently Clark (2009) identified the increasing levels of anthropogenic noise as a habitat concern for whales and other cetaceans because of its potential effect on their ability to communicate. Masking can reduce the range of communication particularly long-range communication, such as that for blue and fin whales. Recent scientific evidence suggests that marine mammals, including blue and fin whales, compensate for masking by changing the frequency, source level, redundancy, or timing of their signals, but the long-term implications of these adjustments are currently unknown (McDonald *et al*. 2006, Parks 2003, 2009).

Surface shipping is the most widespread source of anthropogenic, low frequency (0 to 1,000 Hz) noise in the oceans (Simmonds and Hutchinson 1996). Source levels for commercial ships range from 180-195 dB re 1 µPa which dominate underwater noise in the 10-500 Hz frequency bands (NRC 2003; Hildebrand 2009; McKenna et al. 2012). The Navy estimated that the 60,000 vessels of the world's merchant fleet annually emit low frequency sound into the world's oceans for the equivalent of 21.9 million days, assuming that 80 percent of the merchant ships are at sea at any one time (U.S. Navy 2001). Ross (1976) has estimated that between 1950 and 1975 shipping had caused a rise in ambient ocean noise levels of 10 dB with propeller cavitation primarily responsible for the increase. He predicted that this would increase by another 5 dB by the beginning of the 21st century. NRC (1997) estimated that the background ocean noise level at 100 Hz has been increasing by about 1.5 dB per decade since the advent of propeller-driven ships.

Ambient Noise

Urick (1983) provided a discussion of the ambient noise spectrum expected in the deep ocean. Shipping, seismic activity, and weather are primary causes of deep-water ambient noise (Urick 1983; Hildebrand 2009). Noise levels between 20 and 500 Hz appear to be dominated by distant shipping noise that usually exceeds wind-related noise. Above 300 Hz, the level of wind-related noise might exceed shipping noise. Wind, wave, and precipitation noise originating close to the point of measurement dominate frequencies from 500 to 50,000 Hz. The frequency spectrum and level of ambient noise can be predicted fairly accurately for most deep-water areas based primarily on known shipping traffic density and wind state (wind speed, Beaufort wind force, or sea state) (Urick 1983). For frequencies between 100 and 500 Hz, Urick (1983) has estimated the average deep water ambient noise spectra to be 73 to 80 dB for areas of heavy shipping traffic and high sea states, and 46 to 58 dB for light shipping and calm seas.

In contrast to deep water, ambient noise levels in shallow waters (i.e., coastal areas, bays, harbors, etc.) are subject to wide variations in level and frequency depending on time and location. The primary sources of noise include distant shipping and industrial activities, wind and waves, and marine animals (Urick 1983). At any given time and place, the ambient noise level is a mixture of these noise types. In addition, sound propagation is also affected by the variable shallow water conditions, including the depth, bottom slope, and type of bottom. Where the bottom is reflective, the sound levels tend to be higher than when the bottom is absorptive.

U.S. Navy Activities

The Navy has conducted military training and testing activities throughout the northwest Atlantic Ocean and Gulf of Mexico for decades. In the Atlantic, the U.S. Navy conducts eight types of military training and more than 15 types of military testing activities as well as numerous air operations. Typical training exercises of the Atlantic Fleet include gunnery, missile, surface fire support, Marine expeditionary unit, amphibious assault, bombing, sinking, torpedo, tracking, and mine detection exercises. The Navy undertakes a broad spectrum of testing activities including, but not limited to, basic and applied scientific research and technology development; testing, evaluation, and maintenance of systems (missiles, radar, and sonar), and platforms (surface ships, submarines, and aircraft); and acquisition of systems and platforms to support Navy missions and give a technological edge over adversaries. Testing activities occur in response to emerging science or Navy Fleet operational needs. Many of the testing and training that the Navy routinely undertakes involve the use of both passive and active sonar. Sonar is primarily used by the Navy to detect and defend against submarines (anti-submarine warfare) and mines (mine warfare); safe navigation and effective communications; and oceanographic surveys.

In 1997, NMFS issued a biological opinion on Navy training activities within and in the vicinity of the Atlantic Ocean right whale critical habitat off of the coasts of Georgia and Florida (NMFS 1997). That Opinion concluded that Navy training activities were not likely to jeopardize the continued existence of North Atlantic right whales and other endangered or threatened species or result in the destruction or adverse modification of critical habitat that had been designated in the action area for that consultation.

In the late 1990s, the U.S. Navy implemented several new mitigation measures that were designed to protect right whales. Because of these mitigation measures, NMFS concluded that current Navy operations out of Mayport, Florida were not likely to jeopardize the continued existence of endangered or threatened species under NMFS' jurisdiction (NMFS 1997).

Vessel operations and ordnance detonations adversely affect listed species of sea turtles and whales. U.S. Navy aerial bombing training in the ocean off the southeast U.S. coast involving drops of live ordnance (500 and 1,000-lb bombs) have been estimated to have injured or killed 84 loggerhead, 12 leatherback, and 12 green or Kemp's ridley sea turtles, in combination (NMFS 1997). The Navy ship-shock trials for the USS WINSTON S CHURCHILL were conducted in the Action Area, although the U.S. Navy employed a suite of measures that appeared to protect marine mammals and sea turtles from being exposed to shock waves produced by the underwater detonations associated with the trial (Clarke and Norman 2005).

Between July 2006 and July 2008, the U.S. Navy conducted several Composite Training Unit or Joint Task Force Exercises in and seaward of the Cherry Point and Jacksonville-Charleston Operating Areas. These exercises included antisubmarine warfare training events that employed between 49 and 355 hours of mid-frequency active sonar and deployed between 15 and 170 DICASS sonobuoys. All but two of these exercises were conducted during the summer (the exceptions were a Expeditionary Strike Group Composite Training Unit exercise conducted from mid-May to the first of June 2007 and a Carrier Strike Group Joint Task Force exercise conducted from late April to mid-May 2008), which would have avoided interactions with North Atlantic right whales and most other large cetaceans. The actual number of marine animals that might have been exposed to mid-frequency active sonar during these exercises, and their responses to any exposure, remains unknown; however, no marine animals were reported to have been struck or killed during any of these exercises

In August and September 2008, the U.S. Navy conducted a ship shock trial on the MESA VERDE in waters east of Jacksonville, Florida, using High Blast Explosive (HBX-1) for the detonations (U.S. Navy 2008d). NMFS' biological opinion on the ship shock trial expected up to 36 sea turtles to be injured as a result of the ship shock trial and up to 1,727 turtles to be harassed as a result of their behavioral responses to the underwater detonations. The after action report for the ship shock trial could neither refute nor confirm these estimated number of animals that might have been harassed by the trials; however, surveys associated with the trial did not detect any dead or injured marine mammals or sea turtles during the shock trial event or during post-mitigation monitoring. In addition, no marine mammal or sea turtle stranding events have been attributed to the shock trial.

In 2002, the U.S. Navy established protective measures for North Atlantic right whales for all Atlantic Fleet activities occurring in the Northeast Operating Area. In December 2004, the U.S. Navy issued further guidance for all Fleet ships to increase awareness of right whale migratory patterns and implement additional protective measures along the mid-Atlantic coast, including areas where ships transit between southern New England and northern Florida. The Navy worked with NMFS to identify seasonal patterns of right whale occurrence in six major sections of the mid-Atlantic coast, paying particular attention to port and coastal areas where efforts to manage vessel traffic might be most appropriate. The Navy's resulting guidance directed Navy personnel to exercise extreme caution and operate at slow, safe speeds within 20 nautical mile arcs of specified coastal and port reference points. The guidance reiterated previous instructions that Navy ships post two lookouts, one of whom must have completed marine mammal recognition training, and emphasized the need for utmost vigilance in performance of these watchstander duties.

From 2009 to 2012, NMFS issued a series of biological opinions to the U.S. Navy for training activities occurring within their Northeast, Virginia Capes, Cherry Point and Jacksonville Range Complexes that anticipated annual levels of take of listed species incidental to those training activities through 2014. On June 5, 2009, NMFS issued a biological opinion to the U.S. Navy for training activities occurring within their Northeast, Virginia Capes, Cherry Point and Jacksonville Range Complexes from 2009 through 2010. This biological opinion concluded that while the training activities are not likely to jeopardize species of endangered and threatened whales and sea turtles, there would be instances of harassment and harm. During the proposed activities 2 fin whales, 2 humpback whales, 2 sperm whales, 344 hardshell sea turtles (any combination of green hawksbill, Kemp's ridley or Northwest Atlantic loggerhead sea turtles), 644 Kemp's ridley sea turtles, 21 leatherback sea turtles and 530 NW Atlantic loggerhead sea turtles per year are expected to be harassed as a result of their behavioral responses to mid- and high frequency active sonar transmissions. Another six Kemp's ridley and five Northwest Atlantic loggerhead turtles per year are expected to be injured during exposure to underwater detonations.

On July 28, 2009, NMFS issued a final biological opinion on the U.S. Navy's proposal to place a network of underwater transducer devices and undersea cables in a 1,713-km^2 (500 nautical mile^2) area of the ocean about 93 km (50 nautical miles) offshore of northeastern Florida, beginning in 2012 or 2013 with operations scheduled to begin in 2014 or 2015. The instrumented area, which would be called the Undersea Warfare Tracking Range (USWTR), would be connected by cable to a facility that would be located on shore where the data collected on the range would be used to evaluate the performance of participants in shallow water training exercises.

On June 3, 2010, NMFS issued a biological opinion to the U.S. Navy for training activities occurring within their Northeast, Virginia Capes, Cherry Point and Jacksonville Range Complexes for the years 2010 through 2011. This biological opinion concluded that the training activities are not likely to jeopardize species of endangered and threatened whales and sea turtles. During the proposed activities 2 fin whales, 2 humpback whales, 2 sperm whales, 356 hardshell sea turtles (any combination of green hawksbill, Kemp's ridley or Northwest Atlantic loggerhead sea turtles), 644 Kemp's ridley sea turtles, 21 leatherback sea turtles and 522 NW Atlantic loggerhead sea turtles per year are expected to be harassed as a result of their behavioral responses to mid- and high frequency active sonar transmissions. Another three hardshell, six Kemp's ridley and five Northwest Atlantic loggerhead turtles per year are expected to be injured during exposure to underwater detonations.

In a biological opinion issued on January 6, 2012, for U.S. Navy for training activities occurring within their Northeast, Virginia Capes, Cherry Point and Jacksonville Range Complexes for the years 2011 through 2012, NMFS concluded the training activities are not likely to jeopardize species of endangered and threatened whales and sea turtles. During the proposed activities 2 fin whales, 2 humpback whales, 2 sperm whales, 311 hardshell sea turtles (any combination of green hawksbill, Kemp's ridley or Northwest Atlantic loggerhead sea turtles), 557 Kemp's ridley sea turtles, 20 leatherback sea turtles and 487 NW Atlantic loggerhead sea turtles per year are expected to be harassed as a result of their behavioral responses to mid- and high frequency active sonar transmissions. Another three hardshell, five Kemp's ridley, one leatherback and nine Northwest Atlantic loggerhead turtles per year are expected to be injured during exposure to underwater detonations.

On May 21, 2012, NMFS issued a biological opinion to the U.S. Navy for training activities occurring within their Northeast, Virginia Capes, Cherry Point and Jacksonville Range Complexes for the years 2012 through 2014. This biological opinion concluded that the training activities are not likely to jeopardize species of endangered and threatened whales and sea turtles. During the proposed activities 2 fin whales, 2 humpback whales, 3 sperm whales, 30 hardshell sea turtles (any combination of green hawksbill, Kemp's ridley or Northwest Atlantic loggerhead sea turtles), 555 Kemp's ridley sea turtles, 9 leatherback sea turtles and 466 NW Atlantic loggerhead sea turtles per year are expected to be harassed as a result of their behavioral responses to mid- and high frequency active sonar transmissions. Another three hardshell, five Kemp's ridley, one leatherback and nine Northwest Atlantic loggerhead turtles per year are expected to be injured during exposure to underwater detonations.

Over the same time period (2009 to 2014), the U.S. Navy also conducted mid-frequency active sonar training along the Atlantic coast and the Gulf of Mexico for which NMFS issued biological opinions. Each year during the proposed activities up to 880 blue whales, 970 fin whales, 4,620 humpback whales, 730 North Atlantic right whales, 1,163 sei whales and 10,734 sperm whales are expected to be harassed as a result of their behavioral responses to mid- and high frequency active sonar transmissions and harassed as a result. Because of the frequencies used during the training activities, no sea turtles were expected to be harassed.

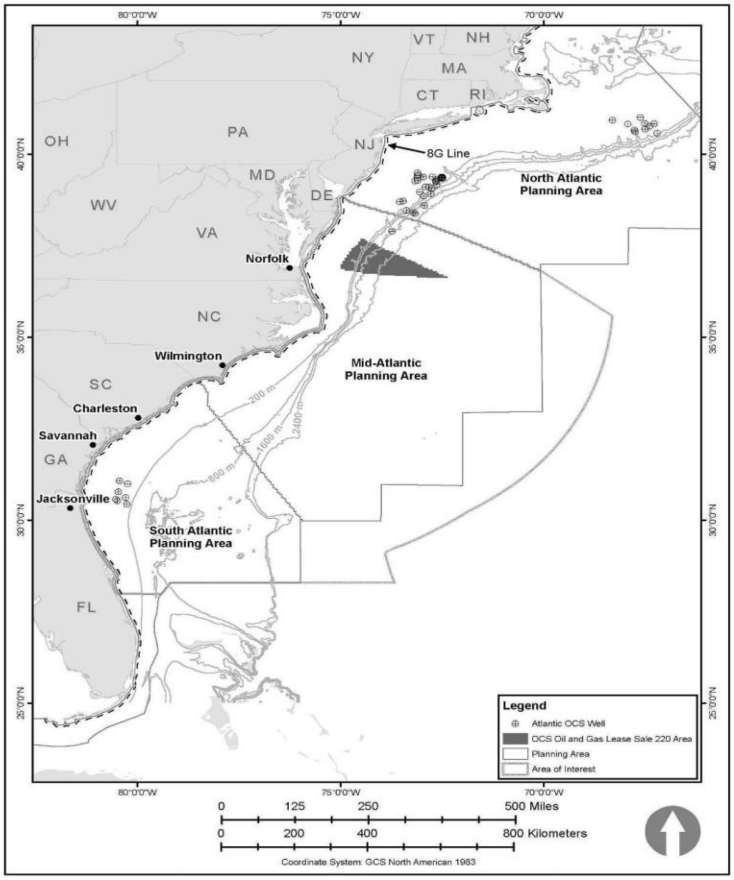
Oil and Gas Activities

Ten oil and gas lease sales were held in the Atlantic between 1976 and 1983. Fifty-one wells were drilled in the Atlantic OCS between 1975 and 1984, including one well in the Mid-Atlantic Planning Area and seven wells in the South Atlantic Planning Area (See Figure 5). Of these 51 wells drilled, five were COST wells drilled between 1975 and 1979. Water depths of these wells ranged from 41 to 819 m (136 to 2,686 ft), and total depth (penetration) ranged from 4,040 to 6,667 m (13,254 to 21,874 ft) (USDOI, BOEM, 2011a *In* BOEM 2012a).

At least one drillable prospect was identified during early Atlantic activity in the 1980s, Manteo, approximately 45 mi (72 km) northeast of Cape Hatteras, North Carolina, in water approximately about 820 m (2,700 ft) deep (USDOI, MMS, 1998). As a result of leasing during Sales 56 and 78 in September 1981 and September 1983, respectively, a 21 OCS block unit was approved by BOEM (then The Minerals Management Service) in 1990 for this prospect.

BOEM's 2007-2012 5-year leasing program for the OCS (MMS 2008) included one sale (Lease Sale 220) offshore the Commonwealth of Virginia, contingent upon the Atlantic drilling moratoria imposed by executive order and the U.S. Congress for the Mid-Atlantic and South Atlantic Planning Areas being lifted at some point within the 5-Year Program. The Program Area for proposed Sale 220 consisted of 593 whole and partial OCS blocks encompassing approximately 2.9 million acres. Water depths in the program area range from 100 to 11,500 ft (30 to 3,500 m). The entire program area was located more than 50 statute miles from the boundary between the Commonwealth of Virginia and Federal waters, and extended seaward to 183 statute miles (Figure 5). Although both moratoria expired and the sale could take place, after the *Deepwater Horizon* event and oil spill in 2010, the Secretary of the Interior re-evaluated the decision to hold Lease Sale 220 and cancelled the sale. There are currently no active oil and gas leases or oil and gas exploration, development, or production activities on the Atlantic OCS.

Figure 5: Locations of 51 Wells Drilled on the Atlantic OCS between 1975 and 1984 and Program Area for Lease Sale 220.



Recovery Actions

Several agencies have engaged in a variety of actions that are designed to reduce the effects of human activities on endangered and threatened species in the Action Area. In 1993, NMFS formed the Southeast Implementation Team for the Right Whale Recovery Plan to address the goals of the Right Whale Recovery Plan within NMFS' Southeast Region. The recovery plan has identified entanglement in fishing gear and ship collisions as the two major direct human impacts affecting both species. Habitat degradation through pollution or other major habitat alteration processes caused by either human sources (discharge or disposal in the marine environment) or resource management activities (fishery or minerals management) is also identified as a major indirect impact requiring attention.

In 1993, the Government of Canada recognized the importance of a portion of the Roseway Basin by designating it as a Right Whale Conservation Area. This basin, which is about 20 nautical miles south of Cape Sable Island Nova Scotia, is one of only two known areas where large numbers of North Atlantic right whales gather on a seasonal basis in Canadian waters.

In 1999, the U.S. Coast Guard implemented a Mandatory Ship Reporting System that requires vessels larger than 300 gross registered tons (Department of the Navy ships are exempt) to report their location, course, speed, and destination upon entering the nursery and feeding areas of the right whale. At the same time, ships receive information on locations of right whale sightings, in order to avoid collisions with the animals. In the southeastern United States, the reporting system is from November 15 through April 15 of each year; the geographical boundaries include coastal waters within roughly 46 kilometers (km) (25 nautical miles [nm]) of shore along a 167 km (90 nm) stretch of the Atlantic coast in Florida and Georgia. In the northeastern United States, the reporting system is year-round and the geographical boundaries include the waters of Cape Cod Bay, Massachusetts Bay, and the Great South Channel east and southeast of Massachusetts; it includes all of Stellwagen Bank National Marine Sanctuary.

An Early Warning System for right whales has been operational in areas of the southeastern U.S. for several years. This system identifies the known location of right whales within and adjacent to the winter calving area from Savannah, Georgia, to Sebastian Inlet, Florida, from 1 December through 31 May (when right whales are assumed to occur in these waters) and provides this information to mariners. This system has successfully diverted shipping to avoid right whales on several occasions, thus decreasing the threat of vessel collisions.

On 1 July 2007, NOAA and the U.S. Coast Guard implemented a shift in the Traffic Separation Scheme servicing Boston to reduce the threat of vessel collisions with right whales and other whale species. The realignment is expected to result in a 58% reduction in the risk of ship strikes to right whales, and an 81% risk reduction in ship strikes of other large whale species occurring in the area.

In 2002, the International Maritime Organization unanimously adopted a Canadian proposal to amend the Bay of Fundy Traffic Separation Scheme to reducing the relative probability of a ship strike in the Roseway Basin by about 80 percent. The Canadian Government proposed establishing a seasonal “Area to be Avoided” in the Roseway Basin, which would apply to ships of 300 gross tonnage or greater, during the seven-month period from June 1 to December 31 when the largest percentage of Right Whales is known to be in the area and when the risk of ship strikes is greatest. The International Maritime Organization’s Maritime Safety Committee adopted Canada's proposal at its 83rd session in Copenhagen Denmark 3-12 October 2007; the newly designated recommended seasonal “Area to be Avoided” took effect six months after it was adopted and was in place prior to the seasonal return of the Right Whales to the Roseway Basin in the spring and summer of 2008.

In October 2008, NMFS established regulations that implement a 10-knot speed restriction for all vessels 65 ft (19.8 m) or longer in certain locations along the east coast of the U.S. Atlantic seaboard at certain times (See Table 7) of the year to reduce the likelihood of deaths and serious injuries to endangered North Atlantic right whales that result from collisions with ships. Evidence suggests that the likelihood of death and serious injury to large whales struck by ships is related to ship speed. The regulations limit ship speed during times and in areas where relatively high right whale and ship densities overlap near a number of U.S. east coast ports, at calving/nursery areas in waters off Georgia and Florida, and in New England waters (See Table 7).

The Impact of the Baseline on Listed Resources

Although 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 as well as 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, the impact of those activities on the status, trend, or the demographic processes of threatened and endangered species remains largely unknown.

Impact on Endangered Whales

Historically, commercial whaling had occurred in the action area and had caused all of the large whales to decline to the point where the whales faced risks of extinction that were high enough to list them as endangered species. Since the end of commercial whaling, the primary threat to these species has been eliminated; however, population sizes of the endangered whales along the Atlantic Coast of the United States still remain at fractions of the population sizes that are estimated to have existed prior to whaling. Nevertheless, populations of species like humpback whales have increased substantially from post-whaling populations levels and appear to be recovering despite the number of individuals that have been killed or injured as a result of ship strikes, interactions with fishing gear, and increased levels of ambient sound along the Atlantic coast. Blue, fin, sei, and sperm whales also exist at smaller population sizes as a result of the legacy of whaling along the Atlantic Ocean, although we know considerably less about the potential effects of many of the stressors associated with the activities considered in this *Environmental Baseline* on growth rates, trend, or age-structure of their populations.

Recent attention has focused on the emergence of a wide number of anthropogenic sound sources in the Action Area and their potential effects on marine animals. Relationships between specific sound sources, or anthropogenic sound generally, and the responses of marine mammals to those sources are still subject to extensive scientific research and public inquiry but no clear patterns have emerged. As a result, the potential consequences of these activities on threatened and endangered marine mammals remain uncertain.

In reviewing studies regarding behavioral responses to human activities, including close approaches by ships, researchers have noted changes in respiration, diving, swimming speed, social exchanges, and other behavior correlated with the number, speed, direction, and proximity of vessels. Responses were different depending on the age, life stage, social status of the whales being observed (i.e., males, cows with calves) and context (feeding, migrating, etc.). Beale and Monaghan (2004) concluded that the significance of disturbance was a function of the distance of humans to the animals, the number of humans making the close approach, and the frequency of the approaches. These results would suggest that the cumulative effects of the various human activities in the Action Area would be greater than the effects of the individual activity. Several investigators reported behavioral responses to close approaches that suggest that individual whales might experience stress responses. Baker *et al*. (1983) described two responses of whales to vessels, including: (1) “horizontal avoidance” of vessels 2,000 to 4,000 meters away characterized by faster swimming and fewer long dives; and (2) “vertical avoidance” of vessels from 0 to 2,000 meters away during which whales swam more slowly, but spent more time submerged. Watkins *et al*. (1981) found that both fin and humpback whales appeared to react to vessel approach by increasing swim speed, exhibiting a startled reaction, and moving away from the vessel with strong fluke motions. Other researchers have noted changes in respiration, diving, swimming speed, social exchanges, and other behavior correlated with the number, speed, direction, and proximity of vessels. Results were different depending on the social status of the whales being observed (single males when compared with cows and calves), but humpback whales generally tried to avoid vessels when the vessels were 0.5 to 1.0 kilometer from the whale. Smaller pods of whales and pods with calves seemed more responsive to approaching vessels (Bauer 1986 and 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 *et al*. 1983, Baker and Herman 1987). 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 (Richardson et. al, 1985; Malme *et al*. 1983). For example, studies of bowhead whales revealed that these whales oriented themselves in relation to a vessel when the engine was on, and exhibited significant avoidance responses when the vessel's engine was turned on even at a distance of about 900 m (3,000 ft). Jahoda *et al*. (2003) studied the response of 25 fin whales in feeding areas in the Ligurian Sea to close approaches by inflatable vessels and to biopsy samples. They concluded that close vessel approaches caused these whales to stop feeding and swim away from the approaching vessel. The whales also tended to reduce the time they spent at surface and increase their blow rates, suggesting an increase in metabolic rates that might indicate a stress response to the approach. In their study, whales that had been disturbed while feeding remained disturbed for hours after the exposure ended. They recommended keeping vessels more than 200 meters from whales and having approaching vessels move at low speeds to reduce visible reactions in these whales.

As we discussed in the *Status of the Species* section of this Opinion, the legacy effects of whaling appear to have had and continue to have greatest effect on endangered Northern Atlantic right whales by reducing them to a population size that is sufficiently small to experience "small population dynamics" (Caughley 1994, Lande 1993, Lande *et al*. 2003, Melbourne and Hastings 2008). At these population sizes, we would expect North Atlantic right whales to have higher probabilities of becoming extinct because of demographic stochasticity, demographic heterogeneity (Coulson *et al*. 2006, Fox *et al*. 2006) —including stochastic sex determination (Lande *et al*. 2003) — and the effects of phenomena interacting with environmental variability. Demographic stochasticity refers to the randomness in the birth or death of an individual in a population, which results in random variation on how many young that individuals produce during their lifetime and when they die. Demographic heterogeneity refers to variation in lifetime reproductive success of individuals in a population (generally, the number of reproductive adults an individual produces over their reproductive lifespan), such that the deaths of different individuals have different effects on the growth or decline of a population (Coulson *et al*. 2006). Stochastic sex determination refers to the randomness in the sex of offspring such that sexual ratios in population fluctuate over time (Melbourne and Hastings 2008).

At small population sizes, populations experience higher extinction probabilities because of their population size, because stochastic sexual determination leaves them with all males or all females (which occurred to the heath hen and dusky seaside sparrow just before they became extinct), or because the loss of individuals with high reproductive success has a disproportionate effect on the rate at which the population declines (Coulson *et al*. 2006). North Atlantic right whales exist at population sizes sufficiently low to experience all or some of these forms of stochasticity and the evidence available suggests that the death of individual females disproportionately increases the rate at which the population declines. Based on the number of other species in similar circumstances that have become extinct (and the small number of species that have avoided extinction in similar circumstances), the longer North Atlantic right whales remain in these circumstances, the greater their extinction probability becomes. We do not yet know to what degree the U.S. and Canadian Traffic Separation Schemes, speed restrictions, and vessel routing activities that NOAA has established along the Atlantic Coast of the United States reduces the number of North Atlantic right whales that are killed or injured during collisions with ships.

The same statement does not appear to be true for blue, fin, humpback, sei, or sperm whales in the Action Area for this consultation, which appear to be increasing in population size -- or, at least, their population sizes do not appear to be declining -- despite their continued exposure to the direct and indirect effects of the activities discussed in the *Environmental Baseline*. Although we do not have information on other measures of the demographic status of these species (for example, age structure, gender ratios, or the distribution of reproductive success) that would facilitate a more robust assessment of the probable impact of the *Environmental Baseline*[2], we infer from their increasing abundance that the *Environmental Baseline* is not currently preventing their population size from increasing.

Impact on Endangered and Threatened Sea Turtles

Several of the categories of activities described in this *Environmental Baseline* have had significant and adverse consequences for nesting aggregations of sea turtles whose individuals occur in the Action Area. In particular, the commercial fisheries that have been described have captured substantial numbers of green, hawksbill, Kemp's ridley, leatherback, and Northwest Atlantic loggerhead sea turtles each year.

---

[2] Increases in a population's abundance is only one piece of evidence that a population is improving in status; however, because populations can increase while experiencing low juvenile survival (for example, if low juvenile survival is coupled with reduced adult mortality) or when those individuals that are most sensitive to a stress regime die, leaving the most resistant individuals, increases in abundance are not necessarily indicative of the long-term viability of a species.

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

Even with TED measures in place, in 2002, NMFS (2002) expected these fisheries to capture about 323,600 sea turtles each year and kill about 5,600 of the turtles captured. Loggerhead sea turtles account for most of this total: 163,000 captured, killing almost 4,000 of them. Kemp's ridleys account for the second-most interactions: 155,503 captures with 4,200 of them dying. These are followed by green sea turtles: about 18,700 captured with more than 500 of them dying as a result of their capture. Leatherback sea turtle interactions were estimated at 3,090 captures with 80 of them dying as a result (NMFS 2002). Since 2002, however, effort in the Atlantic shrimp fisheries has declined from a high of 25,320 trips in 2002 to approximately 13,464 trips in 2009. Since sea turtle takes are directly linked to fishery effort, these takes are expected to decrease but are still expected to result in at least hundreds and possibly in the low thousands of sea turtle interactions annually, of which hundreds are expected to be lethal (NMFS 2012).

Recent data regarding the three largest subpopulations that comprise the Northwest Atlantic loggerhead DPS indicated either that these subpopulations do not show a nesting decline significantly different from zero (Peninsular Florida and The Greater Caribbean subpopulation) or are showing possible signs of nest numbers stabilizing (Northern subpopulation). These trends were declining and have only recently shown signs of stability with strong nesting seasons in the last few years.

Given that we are certain that nest counts of species like Northwest Atlantic loggerhead sea turtles have been declining in the recent past, these additional mortalities each year along with other impacts remain a threat to the survival and recovery of this species. Even if these mortalities did not increase the rate at which certain nesting aggregations are declining, merely continuing the rate at which they are currently declining would be sufficient to increase the probability of nest counts in these nesting aggregations to decline to zero. Because we know that populations of sea turtles cannot increase over time if the number of nest counts decline, the mortalities associated with these fisheries could reverse the estimated progress of Northwest Atlantic loggerheads and increase the probability of populations of green, Kemp's ridley, hawksbill, leatherback or Northwest Atlantic loggerhead sea turtles becoming extinct in the wild.

Impact on Endangered Atlantic Sturgeon

Several activities described in this *Environmental Baseline* have had significant and adverse consequences for Atlantic sturgeon occurring in the Action Area. While commercial fisheries for meat and caviar caused the initial decline for populations within the five DPSs, habitat degradation, coastal runoff and river discharges carrying contaminates remain some of the challenges for Atlantic sturgeon. Atlantic sturgeon are sensitive to pesticides, heavy metals, and other toxins in the aquatic environment. Large sturgeon are most often killed by ship strikes because smaller fish often pass through the propellers without making contact but larger sturgeon get hit. As shipping increases in the future, using larger, deeper draft vessels, as has been predicted, more Atlantic sturgeon are likely to be killed during encounters with ships.

We have been unable to estimate the sizes of the populations for any of the 5 Atlantic sturgeon DPSs; however, it is presumed that the Hudson and Altamaha rivers have the most robust of the remaining U.S. Atlantic sturgeon spawning populations and that the other U.S. spawning populations were likely less than 300 spawning adults per year (ASSRT 2007). At these small population sizes, however, increasing mortality rates or even maintaining current mortality rates is likely to be sufficient to increase the probability of Atlantic sturgeon populations becoming extinct in the wild.

Effects of the Proposed Action

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

Each potential stressor associated with the proposed G&G activities proposed by BOEM is discussed in greater detail below. That section is followed by the results of NMFS' exposure and response analyses, which are designed to determine whether endangered or threatened individuals or designated critical habitat are likely to be exposed to the potential stressor and how those individuals or designated critical habitat are likely to respond, given exposure. This section concludes with an *Integration and Synthesis of Effects* that integrates information presented in the *Status of the Species* and *Environmental Baseline* sections of this Opinion with the results of the exposure and response analyses to estimate the probable risks the proposed action poses to endangered and threatened species.

Stressors Associated with the Proposed Action

The potential stressors we expect to result from the G&G activities proposed by BOEM within the Action Area are represented in Table 9:

Table 9: Stressors Associated with the Proposed Action.

Potential Stressors	Program Area Use			Survey Type(s)
	Oil & Gas	Renewable Energy	Marine Minerals	
Sound Fields Produced by Active Acoustic Sound Sources				
Airguns	X	--	--	Deep penetration seismic surveys and HRG surveys
Electromechanical Sources (subbottom profilers, side-scan sonar, boomers, sparkers, single, swath and multi-beam depth sounders)	X	X	X	HRG surveys for Oil and Gas, Renewable energy and Marine Mineral sites
Vessel and Equipment Noise (including drilling)	X	X	X	All vessel activity; drilling of COST wells and shallow test wells
Vessel Traffic (collisions)	X	X	X	All vessel surveys and support vessel movements
Aircraft Traffic and Noise	X	--	--	Aeromagnetic surveys; helicopter support for COST well drilling
Trash and Debris	X	X	X	Accidental release of trash or debris into the ocean
Seafloor Disturbance				
Bottom Sampling (Temporary placement of cables, nodes, sensors, or anchors on or in seafloor)	X	X	X	Geotechnical sampling and testing
Cables, Nodes, Anchors	X	--	--	Certain deep penetration seismic surveys and CSEM and MT surveys
COST Wells and Shallow Test Drilling	X	--	--	Drilling of COST wells and shallow test wells
Meteorological Buoys	--	X	--	Site characterization for renewable energy areas
Drilling Discharges	X	--	--	Drilling of COST wells and shallow test wells
Accidental Fuel Spills	X	X	X	All vessel surveys

Abbreviations: CSEM = controlled source electromagnetic; COST = Continental Offshore Stratigraphic Test; CPT= cone penetrometer test; HRG = high resolution geophysical; MM = marine minerals; MT=magnetotelluric; OG = oil and gas exploration; RE = renewable energy.

Active Acoustic Sound Sources Associated with the Proposed Action

BOEM anticipates that operators will employ airguns and electromechanical equipment (boomers, sparkers, single, swath and multibeam depth sounders, side-scan sonars and chirp subbottom profilers) in the program areas under the proposed action as reflected in Table 9 above. HRG surveys are often conducted with autonomous underwater vehicles (AUVs) equipped with electromechanical equipment (multibeam depth sounders, side-scan sonars and a chirp subbottom profilers). The following sections give an overview and describe the operating principles of both types of acoustic sources. See Table 10 for all source levels and operating frequencies for all active acoustic sound sources proposed for use over the duration of the proposed action.

Airguns function by rapidly venting high-pressure, compressed air into the water. The pressure signature of an individual airgun consists of a sharp rise and then fall in pressure, followed by several positive and negative pressure oscillations or air bubbles. The best example of this effect is popping a balloon – when the air inside the balloon is quickly released into the surrounding medium (in this case, air), a pressure pulse is created, which is heard as a loud sound (NSF and USDOI, USGS, 2011). An airgun source can consist of a single device, but most often it is made up of an array of airguns. It is considered a low-frequency source since most of its acoustic energy is radiated at frequencies below 200 Hz; however, airgun arrays are broadband emitters, with source spectra spanning a number of third-octave bands from 10 Hz up to 2 kHz (BOEM 2012b). The main specification of an airgun, which defines its broadband source level and spectral (frequency) content, is the volume of the air chamber.

A single airgun is an omni-directional source where the amplitude of the acoustic wave emitted from the source is uniform in all directions. An airgun array, on the other hand, does exhibit directionality because of the varying delays between signals from the spatially separated airguns in different directions. The sizes, arrangement, and firing times of the individual airguns in an array are designed and synchronized to suppress the pressure oscillations (bubbles) subsequent to the pressure release, and coalesce pressure levels into one pulse. The resulting downward-directed pulse has a fraction of a second duration, with only one strong positive and one strong negative peak pressure (Caldwell & Dragoset, 2000).

The source levels for airgun arrays are nominal source levels for sound directed downward, which represent the theoretical source level close to a single point source emitting the same sound as that emitted by the arrays. The actual source for airgun arrays is a distributed sound source (i.e., multiple guns) rather than a single point source. In order to communicate an estimate of the levels of sound from the array, back calculations must be made from far field measurements to acquire a theoretical value of the source level. This theoretical source level is never actually realized for airgun arrays because airguns are distributed over several meters; the highest sound levels actually measurable in the water close to the airgun array (such as one meter) will never be as high as the nominal source level. Given the directional nature (downward) of the sound propagating from these airgun arrays, the effective source level for sound propagating in near-horizontal directions will be substantially lower than the nominal source level, although this horizontal propagation is known to occur over many kilometers from the source. The theoretical point source estimates for airgun arrays are useful, however, for accurately reflecting received levels in the far-field (at more than 75 to 100 m; Caldwell & Dragoset, 2000)

The strengths of airgun pulses can be measured in different ways. Peak-to-peak levels (pk-pk) are presented in units of dB re 1 μPa. The peak level (0-pk) for the same pulse is typically about 6 dB less. In the biological literature, levels of received airgun pulses are often described based on the "average" or "root-mean-square (rms)" level over the duration of the pulse. The rms value for a given pulse is typically about 10 dB lower than 0-pk, and 16 dB lower than pk-pk (McCauley et al., 1998, 2000b; Greene et al. 1997 as cited in NMFS, 2006). A fourth measure – sound exposure level (SEL)– is sometimes used and is expressed in dB re 1 μPa^2•s; however, because seismic pulses are less than one second in duration, the numerical value of the sound energy level is lower than the rms pressure level. Some acoustic experts believe that SEL is a better measure of the received levels of total energy marine animals might experience when exposed to multiple pulses from sources such as seismic airguns and sonar (Southall et al 2007). This measurement becomes less effective in describing the total amount of energy received from multiple airgun pulses, however, since a marine animal would not remain motionless in the sound field, variable environmental characteristics play a role in determining the frequencies and sound levels within a particular sound field and even slight changes in head orientation would affect the amount of energy the animal's auditory system receives.

Pathways for received seismic sound include direct paths from the source, indirect paths that include reflection from the sea surface and bottom, and often indirect paths including segments propagating through bottom sediments. Sound propagating via indirect paths travels longer distances and often arrives later than sounds received via a direct path. However, sound may also travel faster through sediments than in water, and thus may arrive earlier than the direct arrival despite traveling a greater distance. Variations in travel time lengthen the duration of a received pulse; seismic pulses of about 10 to 20 ms in duration at the source can be longer when received at long horizontal distances lessening the impulsive nature of the signal. For example, for an airgun array operating in the Beaufort Sea, pulse duration was about 300 ms at a distance of 8 km, 500 ms at 20 km, and 850 ms at 73 km (C. R. Greene, Jr & Richardson, 1988). Seasonally and spatially variable environmental characteristics also play a role in determining the frequencies and sound levels that a marine mammal may experience when exposed to propagating airgun sounds.

Electromechanical sources can be considered low, mid- or high-frequency emitters. They usually have operating frequencies, which fall in the range from 200 Hz to 900 kHz. The acoustic energy emitted outside the main operating frequency band in most of these devices is negligible; therefore, they can usually be considered narrow band sources. These sources may also be employed simultaneously with airguns during oil and gas exploration.

Boomers are electromechanical low frequency sound sources that generate short, broadband acoustic pulses useful for high-resolution, shallow-penetration sediment profiling. The acoustic impulse from a boomer is generated when two spring-loaded plates are electrically charged causing the plates to repel, thus generating an acoustic pulse. The emitting element of the boomer source is a boomer plate mounted on a catamaran-like sled and towed alongside or astern of the ship. The boomer is a strongly directive source for frequencies above 1 kHz but it becomes omni-directional below 1 kHz.

Sparkers are electrical seismic sources that generate acoustic pulses by vaporizing seawater using high-voltage electrical currents. Sparkers employ large banks of capacitors to generate high voltages, which are then discharged across pairs of underwater electrodes separated by seawater. The spark generated by the electrodes creates steam bubbles in the water. The formation, oscillation, and collapse of these bubbles generate a strongly spiked acoustic pulse in the water that can penetrate several hundred meters into the seafloor.

Chirp subbottom profilers operate at frequencies ranging from 500 Hz to 24 kHz and 200 kHz. The sonar head is mounted at the bottom of the ship's hull, with the central axes of both transducers oriented directly downward. Chirp profilers emit a “swept”-frequency signal, meaning that the transmitted signal is emitted over a period of time and over a set range of frequencies. This repeatable (transmitted) waveform can be varied in terms of pulse length, frequency bandwidth, and phase/amplitude. A matched filter, or correlation process, collapses the swept frequency modulated (FM) received signal into a pulse of short duration, maximizing the signal-to-noise-ratio.

The single, swath, and multi-beam depth sounders emit brief pings of medium- or high-frequency sound (3.5 kHz to 540 kHz) in a single or fan-shaped beam, respectively, extending downward and to the sides of the ship, but not forward or aft. The pulses are sent down toward the ocean floor by a transducer that is mounted to the hull of the ship. In multi-beam echo sounders, the beams produce a fanned arc composed of individual beams, each with a width as narrow as one vertical degree by one horizontal degree. The width of the swath of sonar echo beams produced depends on the water depth as the beams widen as water depth increases. Reflected echoes are received by another array that is mounted parallel to the vessel track, which records the time required for the acoustic signal to travel to and from the seafloor. This number can then be converted into water depth to create a bathymetric (water depth) map.

The side-scan sonar is a dual-frequency, side-scan sonar with two simultaneously-engaged transducers, each producing a full spectrum chirp signal. The sonar can be operated in frequencies ranging from 100 kHz to more than 900 kHz. The sonar is installed inside a streamlined towfish that can be towed behind a vessel at different depths. The central axes of the two transducers are oriented perpendicular to the towing line in the horizontal plane, i.e., at 90^o and 270^o relative to the ship’s course. In the vertical plane, the central axes are tilted downward at 20^o to the horizontal plane. The vertical beam width (across-track) is 50^o for both frequencies. The horizontal beam width (along-track) varies between 0.4^o and 1.26^o, depending on the frequency and the operating mode.

Electromagnetic and Other Active Acoustic Sound Sources

Controlled Source Electromagnetic Surveys (CSEM) surveys used to help delineate potential oil and gas reservoirs induce very low frequency electromagnetic signals into the upper layers of the seafloor via a towed dipole. A vessel tows the dipole (a metal cylinder) that produces, an electrical signal (typically less than 2 hertz [Hz]) that is continuous, not pulsed.

Acoustic Doppler Current Profilers (ACDPs) are employed on met buoys in the renewable energy program can calculate speed of the water current, direction of the current, and the depth in the water column of the current. The ADCP measures water currents with sound, using a principle of sound waves called the “Doppler” effect and works by transmitting high frequency pings of sound at a constant frequency into the water.

Table 10: Representative Acoustic Sound Sources Proposed for Seismic and HRG Surveys in the Oil and Gas, Renewable Energy and Marine Minerals Programs from 2013-2020.

Source	Usage	Broadband Source Level (dB re 1 μPa at 1m)	Pulse Duration	Operating Frequencies
Large Airgun Array (5,400 in^2)	Deep penetration seismic surveys, (2D, 3D, WAZ, VSP, 4D, etc.) Oil and Gas Program	230.7	>100 ms	10 Hz-2,000 Hz (most energy at <200 Hz)
Small Airgun Array (90 in^2)	HRG surveys Oil and Gas Program	210.3	>100 ms	10 Hz-2,000 Hz (most energy at <200 Hz)
Controlled Source Electromagnetic Towed Dipole	CSEM surveys Oil and Gas Program	Not available	N/A	<2Hz
Boomer	HRG surveys All Program Areas	212	180 μs	200 Hz–14 kHz
Sparker	HRG surveys, All Program Areas	210-230	-	50 Hz-500 Hz
Side-Scan Sonar	HRG surveys, All Program Areas	226	20 ms	100 kHz-900 kHz
Chirp Subbottom Profiler	HRG surveys, All Program Areas	222	64 ms	500 Hz to 24 kHz and 200 kHz
Single Beam Depth Sounder	HRG Surveys, All Program Areas	213	>100 ms	3.5 kHz to 540 kHz
Swath Depth Sounder	HRG Surveys, All Program Areas	Not Available	--	100 to 600 kHz
Multibeam Depth Sounder[a]	HRG surveys, All Program Areas	213	>100 ms	70 kHz-500 kHz
ACDP	Renewable Energy Program	Not available	Not available	190 kHz to 2 MHz

[a] Single beam and swath interferometric sounders may also be used for seafloor mapping, and the frequencies and source levels may differ. The multi-beam depth sounder was selected as a representative source for estimating acoustic impacts.

Receiving Devices and Passive Equipment

Hydrophone streamers and geophones act as receiving devices for acoustic sources (i.e., airgun array) during seismic surveys. No acoustic or electromagnetic signals are emitted. The hydrophones or geophones are encased in plastic tubing and either towed behind the survey vessel, positioned on the seafloor as autonomous nodes or cables, or in rare instances spaced at various depths in vertically positioned cables depending on the type of survey conducted.

Magnetotelluric, gravity, gravity gradiometry, and marine magnetic surveys are remote sensing surveys typically conducted from ships or aircraft used to record the Earth's natural electromagnetic field or gravity gradient. No acoustic or electromagnetic signals are emitted from this equipment used during these surveys.

Vessel and Equipment Noise

G&G activities within all three program areas would generate vessel and equipment noise. Most of the vessel activity and noise is associated with 2D and 3D seismic surveys which could occur as early as 2013 and continue until 2020. Other surveys (i.e., WAZ, HRG, CSEM and VSP) would occur at lower levels and within the later years of this proposed action . Vessel and equipment noise would be generated from G&G vessels and their support vessels along with those vessels associated with COST and shallow test well drilling. Broadband source levels for vessels involved in oil and gas activities are anticipated to be in the range of 170-180 dB (re 1 μPa at 1 m) and source levels for boats like those used for renewable energy and marine minerals surveys are in the range of 150-170 dB (re 1 μPa at 1 m) (Richardson et al., 1995b).

Seismic survey vessels could occur anywhere within the Action Area at any time, however, vessels for renewable energy and marine minerals would operate in water depths of 60 m and less and 30 m and less, respectively. Survey vessels for renewable energy and marine minerals projects are expected to make daily round trips to their shore base, whereas the larger seismic vessels can remain offshore for weeks or months. It is anticipated that one to three diesel-powered survey and support vessels would be associated with any specific G&G activity.

Equipment related to drilling activities also produces continuous sounds usually in the low frequency range. BOEM anticipates completion of up to three COST wells and up to five shallow test wells. COST wells are drilled using the same conventional rotary drilling techniques as those used for drilling oil and gas exploration and development wells. Well drilling activities produce strong tonal components at low frequencies (<500 Hz), including infrasonic frequencies in at least some cases (Richardson et al., 1995). Drilling-related noise from jack-up platforms is continuous and generally of very low frequencies (near 5 Hz) with source levels from 119 to 127 dB (re 1 μPa) at nearfield locations. Drilling operations would be supported by crew boats, supply vessels, and helicopters traveling between the drilling rig and the onshore support base.

Drilling from semi-submersible platforms in deeper waters produces an estimated sound source level in the range of 154 (re 1 μPa-m) with frequencies from 10 to 4,000 Hz while source levels for drill ships can be as high as 191 dB re 1 μPa during drilling. At source levels of 154 (re 1 μPa-m) the 120-dB radius would occur at 50 m.

Vessel Collisions

G&G activities within all three program areas would generate vessel and equipment noise. Most of the vessel activity is associated with 2D and 3D seismic surveys which could occur as early as 2013 and continue until 2020. Other surveys (i.e., WAZ, HRG, CSEM and VSP) would occur at lower levels and within the later years for this proposed action than the 2D and 3D surveys. A total of 125 port visits by survey vessels for the oil and gas seismic exploration program would occur during the 2013-2020 period. For the renewable energy program, there would be approximately 3,106-9,969 vessel round trips and approximately 93-615 vessel round trips associated with marine minerals survey activity within the Action Area between 2013 and 2020. These vessels would make port calls at any of the five major ports within the Action Area. Survey vessels typically travel from 5.6 km/hr (3 kts**)** for HRG surveys up to about 11 km/hr (6 kts) for seismic surveying. Vessels traveling to and from ports, however, could travel at speeds considerably more than survey vessels. Vessels deploying and decommissioning bottom-founded monitoring buoys also increase vessel traffic. Seven to 38 buoys are proposed for deployment over the duration of the proposed action. Estimated round trips for deployment and decommissioning is about 64, however, this estimate does not consider maintenance for buoy performance. For post-lease engineering studies under the oil and gas program, AUVs are sometimes used to deploy survey equipment during HRG surveys.

BOEM and BSEE propose to include vessel strike avoidance guidance with all authorizations for shipboard surveys similar to Joint BOEM-BSEE NTL 2012-G01 (BOEM and BSEE 2012b) and described in the Description of the Proposed Action section of this Opinion. This guidance incorporates NMFS Vessel Strike Avoidance Measures and Reporting for Mariners. Shipboard survey operators are also required to abide by NMFS' ship strike reduction rule (50 CFR 224.105) which limits vessel speeds to 10 kts (18 km/hr) in the Mid-Atlantic from November 1 to April 30 annually, and in the Southeast U.S. SMAs from November 15 to April 15, annually, during the North Atlantic right whale migration season. Protected Species Observers (PSOs) on survey ships to alert vessel operators to the presence of marine animals are also expected to help vessels avoid marine mammal strikes. Given the slow speeds associated with the vast majority of vessel transits during surveys, the adherence to NMFS' vessel strike avoidance measures and ship strike reduction rule during right whale migration periods and the presence of PSOs, we expect these measures to be effective in avoiding collisions between vessels and whales in the Action Area.

There is a negligible risk of an AUV striking a marine mammal. Large and/or slow moving species would be more at risk of being struck than smaller, faster swimmers. Most AUVs travel between 5.6 and 14.8 km/hr (3 to 8 kts) which should allow a marine animal sufficient time to avoid the device.

Aircraft Traffic

Surveys using aircraft include aeromagnetic surveys and the installation of COST and shallow test wells. One or two aeromagnetic surveys may be flown at altitudes of between 61 and 152 m (200-500 ft) at speeds of 250 km/hr (135 kts). A typical survey may require 1 to 3 months to complete. Helicopters used to support drilling activities for up to three COST wells and up to five shallow test wells are anticipated to complete one round-trip daily between the drilling rig and the onshore support base. Helicopters are expected to maintain a minimum altitude of 213 m (700 ft) when flying in transit offshore but in practice, offshore support helicopters typically fly at higher altitudes ranging from 229 - 716 m (750-2,350 ft) depending on the distance and direction. COST wells and shallow test wells take approximately 5 to 30 days to complete resulting in a maximum of 30 round trips per well and 450 round trips over the duration of the proposed action. Dominant tones reported for fixed-wing aircraft and helicopters are below 500 Hz generated by not only aircraft engines and propellers but also by the airframe itself (Richardson et al 1995b).

Low-flying aircraft produce sounds that marine mammals can hear when they occur at or near the ocean’s surface. Helicopters generally tend to produce sounds that can be heard at or below the ocean’s surface more than fixed-wing aircraft of similar size and larger aircraft tend to be louder than smaller aircraft. Underwater sounds from aircraft are strongest just below the surface and directly under the aircraft. Aircraft noise is typically present for shorter periods of time and moves at a greater speed due to the higher travel speed of aircraft as oppose to vessel noise (Luksenburg and Parsons 2009). Sounds from aircraft would not have physical effects on marine mammals but represent acoustic stimuli (primarily low-frequency sounds from engines and rotors) that have been reported to affect the behavior of some marine mammals.

There are studies of the responses of marine animals to air traffic and the few that are available have produced mixed results. Some investigators report some responses while others report no responses. Richardson et al. (1995) reported that there is no evidence that single or occasional aircraft flying above large whales in water cause long-term displacement of these mammals.

Several authors have reported that sperm whales did not react to fixed-wing aircraft or helicopters in some circumstances (Au and Perryman 1982, Clarke 1956, Gambell 1968, Green et al. 1992) and reacted in others (Clarke 1956, Fritts et al. 1983, Mullin et al. 1991, Patenaude et al. 2006, Richter et al. 2003, 2006, Smultea et al. 2008, Würsig et al. 1998). Richardson et al. (1985) reported that bowhead whales (*Balaena mysticetus*) responded behaviorally to fixed-wing aircraft that were used in their surveys and research studies when the aircraft were less than 457 meters above sea level; their reactions were uncommon at 457 m and were undetectable above 610 m. They also reported that bowhead whales did not respond behaviorally to helicopter overflights at about 153 m above sea level. Leatherwood et al., (1982) reported that minke whales responded to helicopters at an altitude of 230 m (750 ft) by changing course or slowly diving.

Smultea *et al*. (2008) studied the response of sperm whales to low-altitude (233-269 m) flights by a small fixed-wing airplane off Kauai and reviewed data available from either other studies. They concluded that sperm whales responded behaviorally to aircraft passes in about 12 percent of encounters. All of the reactions consisted of sudden dives and occurred when the aircraft was less than 360 m from the whales (lateral distance). They concluded that the sperm whales had perceived the aircraft as a predatory stimulus and responded with defensive behavior. In at least one case, Smultea and *et al*. (2008) reported that the sperm whales formed a semi-circular “fan” formation that was similar to defensive formations reported by other investigators.

In a review of aircraft noise effects on marine mammals, Luksenburg and Parsons (2009) determined that the sensitivity of whales and dolphins to aircraft noise may depend on the animals’ behavioral state at the time of exposure (e.g. resting, socializing, foraging or travelling) as well as the altitude and lateral distance of the aircraft to the animals. While resting animals seemed to be disturbed the most, low flying aircraft with close lateral distances over shallow water elicited stronger disturbance responses than higher flying aircraft with greater lateral distances over deeper water (Patenaude *et al*. 2002, Smultea *et al*. 2008 *in* Luksenburg and Parsons (2009).

Luksenburg and Parsons (2009) also speculated that areas exposed to repeated and prolonged exposure to aircraft may affect the vocalizations of certain (particularly resident) cetacean populations. Such areas could include coastal airports and whale-watching flight paths as well as areas exposed to daily flights for oil and gas drilling and production.

For sea turtles, sound from low flying aircraft could be heard by a sea turtle at or near the surface based on their sensory biology (Ridgway *et al*. 1969, Lenhardt *et al*. 1994, Bartol 1999, Bartol and Musick 2003, Ketten and Bartol 2006). Sea turtles might also detect low flying aircraft via visual cues such as the aircraft's shadow. Hazel *et al*. (2007) suggested that green sea turtles rely more on visual cues than auditory cues when reacting to approaching water vessels. This suggests that sea turtles might not respond to aircraft overflights based on noise alone. As such, sea turtles may react to fixed-wing aircraft and helicopter overflights only as they can visually detect them. We assume that sea turtles would react to overflights in much the same ways as marine mammals once an aircraft is detected.

The effects of aircraft noise on fish include startle, and possibly stress, responses to low flying aircraft. However, given the limited coastal distribution of Atlantic sturgeon (within 50 m water depths) and wide dispersal along the coast, we do not expect more than occasional exposure of these fish to aircraft overflights, if they are exposed at all.

Richardson *et al*. (1995b) reported that there is no evidence that single or occasional aircraft flying above large whales in water cause long-term displacement of these mammals while Luksenburg and Parsons (2009) concluded that marine mammals exposed to aircraft overflights may exhibit no response or behavioral reactions such as quick diving. Any behavioral avoidance reaction for marine mammals, sea turtles or Atlantic sturgeon would be short-term and would not permanently displace animals or result in physical harm. Given this, and the low number of aircraft overflights over the duration of the proposed action we conclude that any responses of marine mammals, sea turtles or Atlantic sturgeon to disturbance associated with aircraft is likely to be short-lived and will not rise to the level of harassment. As such we will not consider aircraft disturbance further in this Opinion.

Seafloor Disturbance

Seafloor disturbance can occur from sediment sampling and placement and removal of equipment on the seafloor during oil and gas, renewable energy and marine minerals program activities. Sampling includes cone penetrometer tests, geologic coring, grab sampling and vibracoring and individual sampling events lasts three days or less. Seafloor disturbance and scour can also occur from bottom founded monitoring buoy anchors and anchors associated with VSP and ocean bottom cable surveys. Anchors associated with monitoring buoys, VSP and ocean bottom cable surveys are left in place after buoys are decommissioned or the survey has ended and are biodegradable.

Sampling for renewable energy projects would occur at specific sites (consisting of one or more OCS blocks) in water depths less than 100 m (328 ft) with most occurring in water depths less than 60 m, and along potential cable routes to shore. Offshore Delaware, Maryland, and Virginia, the likely sampling locations would be within designated WEAs. Designated WEAs in the Mid-Atlantic Planning Area range in size from 164 nmi^2 (138,788 ac) offshore Virginia, to 94 nmi^2 (79,706 ac) offshore Maryland. North Carolina has identified 354 OCS blocks of interest, but it is likely that sampling would occur within only a small subset of these blocks. Specific WEAs have not been identified for the South Atlantic Planning Area.

Sampling activities for marine minerals would be conducted at specific borrow areas in water depths less than 30 m (98 ft). Much of the marine minerals activity is expected to occur within existing borrow areas within the mid-Atlantic and south Atlantic Planning Areas although new areas may be identified and used over the duration of this proposed action.

Sampling for oil and gas exploration would be conducted at specific lease blocks where structures (e.g., drilling rigs, platforms, floating production structures, or pipelines) may be installed. The blocks could be anywhere within the Mid- or South Atlantic Planning Areas. However, requirements for mapping and avoidance, as well as pre-deployment photographic surveys of areas where bottom-founded instrumentation and appurtenances are to be deployed, would be required for all surveys.

Bottom sampling activities would primarily take place in soft bottom areas as most bottom sampling equipment cannot penetrate hard bottom substrate. Piston and gravity cores are approximately 8-cm (3-in) diameter holes in the seafloor and, depending upon the firmness of the seafloor, the core or probe weight stand (30-45 cm [12-18 in] diameter footprint) may also impact the seafloor. Grab sampling is performed to identify the benthic fauna and penetrates from a few inches to a few feet below the seafloor and typically involves 30-40 grabs within an area of interest. Piezocone penetrometer testing is performed to determine the sediment engineering properties and uses a probe typically 10-1,500 mm^2 in diameter, mounted on a frame, and lowered to the seafloor. A vibracore survey generally uses a 7-cm (2.8-in) diameter core barrel mounted on a 2- to 4-m^2 platform and that can penetrate sediments between 6 -15 m (20-50 ft) below the seafloor. Fifteen to twenty five cores would be obtained in a 1 mi^2 (259 ha) area of interest.

Total bottom sampling activities includes 50-300 core or grab samples in the oil and gas program; 6,914-22,209 core samples and 6,914-22,209 grab samples in the renewable energy program; and 1-8 geologic cores, 60-320 grab samples, and 90-600 vibracores in the marine minerals program. Collection of each sample is estimated to disturb an area of approximately 10 m^2, although the actual area of the core or grab extracted may be much smaller.

The total area of seafloor disturbed by bottom sampling and shallow coring activities is estimated to be a very small area (0.000005 percent of the Action Area) both individually and cumulatively within the Action Area (85,477,900 ha, (211,220,480 ac). Although several thousand cores may be collected under the renewable energy program, sampling in soft bottom areas would produce only minor, localized turbidity which is expected dissipate when sampling ends.

The installation of COST wells and shallow test wells has the potential to impact benthic resources. For this impact analysis, the area of seafloor disturbance is assumed to average about 2 ha (5 ac) per well. If all of the COST wells and shallow test wells in the proposed action scenario were drilled, the total seafloor disturbance would be about 16 ha (40 ac), or about 0.00002 percent of the AOI. Requirements for mapping and avoidance, as well as pre-deployment photographic surveys of areas where bottom-founded appurtenances are to be deployed, would be required.

Up to 16 ha (40 ac) of soft bottom could be impacted from the installation of COST wells and shallow test wells. About 0.00002 percent of the Action Area can be impacted by well drilling with each well disturbing up to 2 ha (5 ac) of soft bottom. This disturbance could result in changes in species composition, community structure, and/or ecological functioning of these soft bottom areas.

BOEM would implement requirements to ensure protection of sensitive benthic resources, including setbacks from sensitive bottom communities similar to those currently required for the Gulf of Mexico for activities within the Action Area. Given that BOEM would require prior approval of any G&G activities involving seafloor-disturbing activities or placement of bottom-founded equipment or structures (including requiring site-specific information), sensitive benthic resources would be avoided.

Under the renewable energy program between 7 and 38 bottom-founded monitoring buoys could be installed over the duration of the proposed action. Similarly, the placement and removal of bottom cables and anchors would produce localized sediment disturbance to soft bottom communities. Anchors for either boat-shaped or discus-shaped buoys are expected to produce a footprint of about 0.55 m^2 (6 ft^2) and an anchor sweep of about 3.4 ha (8.5 ac). Total footprint area would range from 3.8 to 20.9 m^2 (42 to 228 ft^2) for the anchors, while the sweep area would range from 23.8 to 129.2 ha (59.5 to 323 ac).

The maximum, total seafloor disturbance would be about 129 ha (319 ac), or about 0.0002 percent of the AOI and 0.047 percent of the identified WEAs if all buoys were installed. The impact from anchoring and anchor sweep would be spread out over the identified WEAs, as typically only one to two buoys are installed per lease. An average lease is approximately 10 lease blocks.

Each individual area of impact from a buoy anchor would be small (approximately 0.55 m^2 [6 ft^2] each), and the anchor sweep area (3.4 ha [8.5 ac] per buoy) comprises 0.015 percent of an average lease area. In addition, the anchor sweep impacts are caused by the anchor chains or lines sweeping the soft bottom substrate; sediments are not removed, and only the surficial sediments and associated soft bottom community are impacted, not the soft bottom community present beneath the surficial sediments (BOEM 2012b). No overall changes in species composition, community structure, and/or ecological functioning of soft bottom communities are expected (Grannis, 2005).

Only localized turbidity is expected to occur as a result of the proposed bottom sampling activities. This turbidity is expected to dissipate after sampling activities have ceased. Anchors associated with monitoring buoys and VSP and ocean bottom cable surveys are left in place after buoys are decommissioned or the survey has ended and are biodegradable. Based on the above, we would not expect adverse effects to listed species from bottom sampling activities and we will not consider bottom disturbance further in this Opinion.

Trash and Debris

G&G survey operations and drilling rigs used during COST and shallow test wells generate trash comprised of paper, plastic, wood, glass, and metal mostly from galley and offshore food service operations. A substantial amount of waste products could be generated from G&G activities over the duration of the proposed action. Because G&G activities can occur anywhere within the

Action Area (within state waters out to the extended continental shelf), the possibility exists that trash and debris could be released into the marine environment.

All survey vessels performing work within U.S. jurisdictional waters are expected to comply with Federal regulations that implement the International Convention for the Prevention of Pollution from Ships (MARPOL) as amended by the 1978 Protocol (MARPOL 73/78). Within MARPOL Annex V, Regulations for the Control of Pollution by Garbage from Ships, as implemented by 33 CFR 151, are requirements designed to protect the marine environment from various types of garbage generated on board vessels. These requirements include, a prohibition on the deliberate discharge of containers and other similar materials (i.e., trash and debris) into the marine environment unless it is passed through a comminutor that breaks up solids and can pass through a 25-mm mesh screen; a prohibition on the discharge of plastic regardless of size; markings on equipment, tools and containers (especially drums), and other material as well as recording and reporting of items lost overboard, and; precautions for handling and disposing of small items and packaging materials.

In addition to MARPOL requirements, BSEE requires that all vessel operators, employees and contractors actively engaged in G&G surveys be briefed on marine trash and debris awareness elimination as described in BSEE NTL No. 2012-G01 (*Marine Trash and Debris Awareness and Elimination*). This NTL (BOEM and BSEE 2012) requires operators to ensure that its employees and contractors are made aware of the environmental and socioeconomic impacts associated with marine trash and debris and their responsibilities for ensuring that trash and debris are not intentionally or accidentally discharged into the marine environment.

Because operators must comply with Federal regulations, BOEM’s trash and debris guidance, the vastness of the Action Area, the amount of trash and debris occurring within the Action Area is expected to be minimal and distributed over a wide area. As such we do not expect exposure of listed species to trash and debris and will not consider this further in this Opinion.

Drilling Discharges

Drilling discharges consist of drilling fluids used during well drilling and drill cuttings from seafloor bedrock. Water-based drilling fluids (WBFs) are used in shallow wells (shallow test wells) or the shallow segment of deeper wells (COST wells) while synthetic-based drilling fluids (SBFs) are used in deeper wells (COST wells). Although COST wells or shallow test drilling activity is infrequent, they are expected in the Blake Plateau region of the South Atlantic Planning

Area. A deep stratigraphic test well is defined as a well of at least 152 m (500 ft) in depth; otherwise it is classified as a shallow test well.

During drilling, fluid and cuttings would be discharged, disperse in the water column, and accumulate on the seafloor around the well. Synthetic-based drilling fluids s are recycled and are not discharged directly; however, some fluid is retained on cutting particles that are discharged and deposited on the seafloor at and near the wellsite. Assuming an average of 2,000 bbl of cuttings and 8,350 bbl of drilling fluid discharged per well, the total volume for one to three COST wells would range from 2,000 to 6,000 bbl of cuttings and 8,350 to 25,050 bbl of drilling fluid (U.S. DOI, BOEM 2012a).

In general, cuttings with adhering SBFs tend to clump together and form piles close to the drill site. SBF cuttings may cause elevated levels of organic carbon concentrations resulting in anoxic conditions (Continental Shelf Associates, Inc., 2006). Where cuttings have accumulated as a result of drilling multiple wells in proximity to one another, concentrations of organic carbon have exceeded approximately 1,000 mg/kg. In these instances, benthic infaunal communities have been affected by drilling fluid toxicity and organic enrichment that can cause anoxia (Neff et al., 2000). However, within 3-5 years after SBF discharges, a complete recovery of the benthic community is possible (Neff et al., 2000). The areal extent of impacts from drilling discharges during the proposed action would be small. Assuming a typical effect radius of 500 m (1,640 ft), the affected area around each wellsite would represent about 3 percent of the seafloor within an OCS lease block. Because shallow test wells and COST wells are expected to occur in the deeper waters of the South Atlantic Planning Area, the small radius of the area affected by drilling fluids and cuttings and because of the limited number of proposed wells, (0-5 hallow test wells and 0-3 COST wells) we expect that any toxic effects associated with SBFs and cuttings will be limited to a small area (500 m) of an OCS block. Given the small area affected within the Action Area we do not expect adverse effects to listed species from drilling discharges, and we will not consider drilling discharges further in this Opinion.

Accidental Fuel Spills

There are few drilling events planned over the duration of the proposed action. Although up to three COST wells and up to five shallow test wells will be drilled, these wells will not be drilled in areas where there are suspected reservoirs of oil or gas. Any accidental fuel spills that may result from G&G activities will result from accidental release of fuel or diesel from survey vessel groundings or other accidents. A spill could occur anywhere within the Action Area as G&G surveys could occur anywhere within the Action Area. While considered relatively rare events, spills when they occur can result in acute effects of short duration and limited impact, or they may have long-term population- or ecosystem-level impacts depending on the timing, spill size and duration and the numbers and types of organisms affected (NRC 2003).

Vessel fuel capacities generally depend on vessel size, which varies according to the nature of the survey (for example, 3D surveys use larger vessels than 2D surveys). A large seismic survey vessel may carry between 100,000-1.1 million gal (2,380-27,000 barrels, bbl) of fuel, including diesel and fuel oil (CGGVeritas, 2011; Geophysical Service, Inc., 2011a,b). Smaller coastal vessels may carry several thousand gallons. Spill size would depend on the type of vessel, the severity of the event, and whether the fuel storage is compartmentalized.

BOEM evaluated a spill scenario for a release of 1.2-7.1 bbl of diesel fuel caused by either a vessel collision or an accident during fuel transfer. The volume is based on spill statistics for the period 2000-2009 developed by the USDHS, USCG (2011b *in* DOI, BOEM 2012a). During this period, there were 1,521-5,220 spills per year from vessels other than tankers and tank barges. Total annual spill volumes from these vessels ranged from 92,388-453,901 gal, resulting in average spill sizes ranging between 49.6 and 297.3 gal, or 1.2-7.1 bbl.

Spilled oil, including refined diesel, can lead to a range of effects including brain lesions, stress, and disorientation from inhalation (NRC 2003) to modification of swimming speed and direction or reduced surface time in oiled waters with no obvious ill effects (Geraci 1990). Marine mammals and sea turtles surfacing within or near a diesel release would be expected to inhale petroleum vapors, causing respiratory stress. Transfer of hydrocarbons through the food chain is initiated by single-cell organisms, such as phytoplankton, which are exposed to hydrocarbons primarily through attachment to organic-rich particles, including plankton and detritus and which are ingested by higher trophic levels (i.e., filter feeders, fish, and mammals). Ingested diesel fuel, particularly the lighter fractions, can be acutely toxic to marine species. The extent of hydrocarbon levels within an organism is controlled by the breakdown rate of the hydrocarbons in the organism after ingestion (NRC 2003). Oil on the skin and shell of a marine turtle can affect respiration, salt gland functions, digestive and immune systems, and blood chemistry (NOS, 2003). Oil exposure can increase egg mortality and lead to developmental defects, as well as cause direct mortality in hatchlings, juveniles, and adults (NOS, 2003). Although major oil spills are relatively rare events, chronic exposure to low levels of oil may impair a sea turtle's overall level of fitness and reduce its ability to withstand other stresses in the environment (NOS, 2003).

Effects of spilled oil on fishes have been studied extensively (Hose et al., 1996; Kocan et al., 1996; Carls et al., 1999; Couillard et al., 2005; Ramachandran, 2005; Schein et al., 2009 *in* U.S. DOI, BOEM 2012a). Schein et al. (2009 *in* U.S. DOI, BOEM 2011a) showed that the constituents of diesel remaining 18 hr after simulated weathering and solubilization were chronically toxic to rainbow trout. Additional studies have shown significant reduction in growth of embryos, a cessation of development, or both, as indicated by a smaller size and reduced absorption of yolk. Release and dissolution of spilled diesel may also decrease ambient oxygen concentrations in the water column.

Only subadult and adult Atlantic sturgeon are expected to occur within the Action Area for this consultation as juvenile and young-of-the-year sturgeon habitat occurs within their natal rivers well away from the influence of oil spills further offshore. Atlantic sturgeon are demersal and are unlikely to be contacted by a diesel spill which is expected to float and disperse on the sea surface. There is the potential for a small proportion of the heavier fuel components to adhere to particulate matter in the upper portion of the water column and sink. However, due to the assumed small size of the spill, it is unlikely that benthic habitats would be contaminated to an extent that would significantly affect any Atlantic sturgeon.

In many oil spills, evaporation is the most important process in terms of amount of oil remaining in the water column over time. Oxygen concentrations, nutrients, water temperature, salinity, the physical properties of the oil and the energy level of the environment all play a role in biodegradation rates of oil in water (NRC 2003, 2005). Diesel fuel is a refined petroleum product that is lighter than water. It may float on the water's surface or be dispersed into the water column by waves. It is assumed that spilled fuel would rapidly spread to a layer of varying thickness and break up into narrow bands or windrows parallel to the wind direction. This process is expected to increase the weathering and breakdown of the fuel. Diesel is a distillate of crude oil and does not contain the heavier components that contribute to crude oil's longer persistence in the environment. Diesel fuel is a refined petroleum product that is lighter than water. It may float on the water's surface or be dispersed into the water column by waves. An accidental diesel fuel spill from a G&G survey vessel would be expected to disperse quickly in the open ocean with small diesel spills (500-5,000 gal) typically evaporating and dispersing within a day (U.S. DOC, NOAA, 2006). Marine diesel, however, is often a heavier intermediate fuel oil that will persist longer when spilled. When spilled on water, diesel oil spreads very quickly to a thin film of rainbow and silver sheens, except for marine diesel, which may form a thicker film of dull or dark colors (U.S., DOC, NOAA, 2006). There is the potential for a small proportion of the heavier fuel components to adhere to particulate matter in the upper portion of the water column and sink.

The only wells proposed within this proposed action are COST and shallow test wells that are drilled away from suspected oil and gas reserves. Oil spills that could occur would be those resulting from accidental release of oil or fuel from vessel operations or collisions. All G&G vessels are required to comply with USCG requirements relating to prevention and control of oil spills and the likelihood of a fuel spill during seismic surveys or other G&G activities is expected to be remote. There has never been a recorded oil/fuel spill during more than 54,000 nmi (100,000 km) of previous NSF-funded seismic surveys (NSF and USDOI, USGS, 2011). Because of the low expected likelihood of a spill, the small extent of any accidental spill, the weathering process of spilled oil and the vastness of the Action Area, we do not expected listed species to be adversely affected by a spill; therefore we will not consider accidental fuel spills further in this Opinion.

Exposure Analysis

Marine Mammals

Endangered whales occur throughout the Action Area for this. As the specific spatial and temporal aspects of individual G&G activities are unknown at this time, we assume, based on the best scientific and commercial data available, that G&G activities can co-occur with endangered whales in the Action Area as indicated below.

Blue Whales. In the western North Atlantic Ocean, blue whales are found from the Arctic to at least the mid-latitude waters of the North Atlantic (CeTAP 1982, Wenzel *et al.*1988, Yochem and Leatherwood 1985, Gagnon and Clark 1993). Yochem and Leatherwood (1985) suggested an occurrence of this species south to Florida and the Gulf of Mexico. Blue whales have been observed frequently off eastern Canada, particularly in waters off Newfoundland, during the winter. In the summer months, they have been observed in Davis Strait (Mansfield 1985), the Gulf of St. Lawrence (from the north shore of the St. Lawrence River estuary to the Strait of Belle Isle), and off eastern Nova Scotia (Sears *et al.* 1987).

Fin Whales. Fin whales are common off the Atlantic coast of the United States in waters immediately off the coast seaward to the continental shelf (about the 1,000-fathom contour) but are less concentrated in nearshore environments. In this region, they tend to occur north of Cape Hatteras, where they accounted for about 46 percent of the large whales observed in surveys conducted between 1978 and 1982. During the summer months, fin whales in this region tend to congregate in feeding areas between 41°20'N and 51°00'N, from shore seaward to the 1,000-fathom contour. Clark (1995) reported a general southward pattern of fin whale migration in the fall from the Labrador and Newfoundland region, south past Bermuda, and into the West Indies.

Humpback Whales. Humpback whales migrate seasonally between warmer, tropical or sub-tropical waters in winter months (where they reproduce and give birth to calves) and cooler, temperate or sub-Arctic waters in summer months (where they feed). During seasonal migrations, humpback whales disperse widely in deep, pelagic waters and tend to avoid shallower coastal waters (Winn and Reichley 1985). Most humpback whales in the western North Atlantic Ocean migrate to the West Indies to mate (e.g., Dominican Republic); however, some whales do not make the annual winter migration (Waring et al., 2010). Sightings data show that humpback whales traverse through coastal waters of the southeastern U.S., including the Action Area (Waring et al., 2010). Swingle et al. (1993) and Barco et al. (2002) reported humpback sightings off Delaware Bay and Chesapeake Bay during the winter, which suggests the mid-Atlantic region may also serve as wintering grounds for some Atlantic humpback whales. This region has also been suggested as important area for juvenile humpbacks (Wiley et al., 1995).

North Atlantic Right Whales. North Atlantic right whales are most abundant in Cape Cod Bay between February and April (Hamilton and Mayo 1990 Schevill *et al*. 1986, Watkins and Schevill 1982), in the Great South Channel in May and June (Kenney *et al.* 1986, Payne *et al*. 1990), and off Georgia and Florida from mid-November through March (Slay *et al*. 1996). Right whales also use mid-Atlantic waters as a migratory pathway between the winter calving grounds and their spring and summer nursery and feeding areas in the Gulf of Maine.

Sei Whales. Sei whales are often associated with deeper waters and areas along the continental shelf edge (Hain *et al.* 1985); however, this general offshore pattern of sei whale distribution is disrupted during occasional incursions into more shallow and inshore waters (Waring *et al.* 2004). In the western Atlantic Ocean, sei whales occur from Labrador, Nova Scotia, and Labrador in the summer months and migrate south to Florida and the northern Caribbean in the winter months (Gambell 1985, Mead 1977).

Sperm Whales. In the western Atlantic Ocean, sperm whales are distributed in a distinct seasonal cycle, concentrated east-northeast of Cape Hatteras in winter and shifting northward in spring when whales are found throughout the Mid-Atlantic Bight. Distribution extends further northward to areas north of Georges Bank and the Northeast Channel region in summer and then south of New England in fall, back to the Mid-Atlantic Bight. Sperm whales have a strong preference for the shelf break and seaward but have been observed in water between 41-55 meters deep; therefore sperm whales could be exposed to seismic surveys and HRG surveys.

Exposure to Seismic and HRG Surveys

The narratives that follow present the results of the approach the BOEM used to estimate the number of marine mammals that might be "taken" (as that term is defined pursuant to the MMPA) during seismic and electromechanical survey activities proposed for 2013-2020 within and adjacent to the Mid- and South Atlantic Planning Areas (which are also described in the *Approach to the Assessment* section of this Opinion) and the results of our independent analyses of BOEM's estimates.

Marine Mammal Take Estimates

In the past, the geophysical community and NOAA Fisheries have estimated the potential impacts of exposing marine mammals to airguns employed during seismic surveys by (1) determining the estimated distance from a source to a particular isopleth that corresponded to harassment under the MMPA (the 160 dB received level for Level B harassment and the 180 dB received levels for Level A harassment of cetaceans); (2) assume that a source propagated cylindrically to estimate the range to these isopleths; (3) calculate the surface area ensonified by this cylinder as the source moved along its track; and (4) multiply the resulting surface area by the density of each marine mammal species present to estimate the number of the different species that would be "taken" through Level A and Level B harassment (as those terms are defined pursuant to the MMPA).

In their Programmatic EIS, BOEM used a different approach to estimate the number of the different species that would be "taken" through Level A and Level B harassment (as those terms are defined pursuant to the MMPA). First, BOEM modeled the different sources and their properties, the acoustic propagation field in 3D, and 3D animal placement and movement to better calculate the potential impacts to marine mammals. The first step of this methodology - propagation - is largely controlled by properties of the source, such as its movement in time and space, the sound field it generates at any point in time, and the effect of the physical properties of the ocean on the sound field as it propagates out from the source. The second step of BOEM's approach relied on the Acoustic Integration Model (AIM), which integrates acoustic exposures over time and exposed virtual representations of marine animals to the results of the integrated acoustic exposures, to estimate the number of the different species that would be "taken" through Level A and Level B harassment (as those terms are defined pursuant to the MMPA).

Although this methodology produces more precise sound fields than approaches that have been used in the past, the virtual animals it relies on do not closely approximate the behavior of real, free-ranging animals. Specifically, the model oversimplifies animal hearing, animal movement (the virtual animals primarily move vertically in the water column and do not actively or passively avoid a sound field), and animal behavior (in addition to not trying to avoid a sound field, the model assumes that the virtual animals are not exposed to other competing stimuli, that their behavior is unaffected by experience or physiological state, etc.). More importantly, the model assumes that seismic surveys primarily affect marine animals by affecting their hearing sensitivity rather than through their behavioral responses to acoustic exposures.

However, acoustic stimuli can represent two different kinds of stressors: a *systemic stressor*, which usually elicit direct physical or physiological responses and, therefore, do not require high-level cognitive processing of sensory information and a *processive stressor*, which require high-level cognitive processing of sensory information (Anisman and Merali 1999, de Kloet *et al*. 2005, Herman and Cullinan 1997, Johnson 2011). Our concern for noise produced by seismic surveys as systemic stressors would lead us to focus on effects such as permanent and temporary threshold shift and acoustic resonance. Our concern for those noises as processive stressors would lead us to focus on whether an animal is likely to attend to the sound (as opposed to competing sounds), how an animal categorizes the sound, the behavior of the signal's source, the animal's behavioral state, and motivations, among other variables. The approach BOEM used in their modeling analyses focuses on sound as a systemic stressor only and not its role as a processive stressor.

BOEM estimated the number of endangered and threatened species that might be "taken," as that term is defined pursuant to the MMPA, not the number of those species that might be exposed to sounds produced by seismic and HRG survey activities. Obviously, an animal that is "taken" by these sounds must first be exposed to the sounds; however, not every animal that is exposed responds in a way that would constitute "take" regardless of how we define that term (for example, many animals exposed at low received levels are not likely to respond to those sounds).

The MMPA defines "harassment" as "any act of pursuit, torment, or annoyance which has the potential to injure a marine mammal or marine mammal stock in the wild or has the potential to disturb a marine mammal or marine mammal stock in the wild by causing disruption of behavioral patterns, including, but not limited to, migration, breathing, nursing, breeding, feeding, or sheltering" [16 U.S.C. 1362(18)(A)]. Level A harassment is defined as "any act that injures or has the significant potential to injure a marine mammal or marine mammal stock in the wild." Level B harassment is defined as "any act that disturbs or is likely to disturb a marine mammal or marine mammal stock in the wild by causing disruption of natural behavioral patterns, including, but not limited to, migration, surfacing, nursing, breeding, feeding, or sheltering, to a point where such behavioral patterns are abandoned or significantly altered."

The ESA defines "take" as harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect or attempt to engage in any such conduct" (16 USC 1532(19)). NOAA Fisheries has not promulgated a regulatory definition of "harass"; however, it has promulgated regulations that further define "harm" to mean "an act which actually kills or injures fish or wildlife. Such an act may include significant habitat modification or degradation which actually kills or injures fish or wildlife by significantly impairing essential behavioral patterns, including, breeding, spawning, rearing, migrating, feeding or sheltering" (50 CFR 222.102).

We use BOEM's estimates as a rough estimate of the number of instances in which endangered and threatened species might be exposed to and "taken" by sounds produced by seismic and HRG survey activities along the Atlantic seaboard. However, these "take" estimates will underestimate the number of instances in which these species would have been exposed to those sounds and would have responded in ways that we would consider "adverse" even if those adverse responses did not constitute one or more forms of "take."

The number of whales BOEM estimates would be "taken" during the G&G activities BOEM plans to authorize off the Atlantic seaboard from 2013 through 2020 depends on the acoustic criteria being used. (1) NMFS' current 180 dB criterion for "level A harassment" under the MMPA where a listed whale within a 180 dB radial distance from an acoustic source has a potential to be injured as a result of their exposure to the acoustic source or Southall et al's (2007) proposed criteria for level A harassment", which is based on information regarding the lowest received levels of impulsive sounds (e.g., airgun pulses) that might elicit slight auditory injury. These received levels are measured by a cumulative energy-based metric, sound exposure level (SEL), of 198 dB re 1μPa^2-s and an instantaneous peak sound pressure level exceeding 230 dB re

1 μPa (flat-weighted). Exceeding either criterion may potentially elicit auditory injury. NMFS' criterion for "level B harassment" defines the radial distance from an acoustic source that potentially induces behavioral harassment.

NMFS is currently updating and revising all of its acoustic criteria and thresholds. Since that process in not complete, for this Opinion we will consider NMFS' 160 and 180 dB re 1μPa thresholds during our analysis. We recognize, however, that individual marine mammals will change their behavioral state in ways that may constitute harassment at received levels that are higher and lower than 160 re 1μPa and we discuss these changes in our response analysis.

BOEM's take estimates represent the annual numbers of individuals taken. For each year, these estimates were calculated to three decimal places, and values were rounded to the nearest whole number of animals. The total number of takes is the sum of the rounded annual values. NMFS, in using BOEM's take estimates, estimated mean values and confidence intervals based on the t-distribution and using the procedures and formulae described by Cumming (2012). NMFS estimated mean values and confidence intervals associated with annual as well as cumulative (through 2020) takes for each species of whale. In some instances, NMFS' take estimates differed from BOEM's take estimates. BOEM's original proposed action included seismic and HRG surveys in 2012, however, BOEM plans to redistribute 2012 effort between 2013 and 2020, although it is not known where or when surveys will occur. Estimated takes have also been redistributed but within any year, takes should remain within confidence intervals. BOEM estimated takes for G&G surveys in two categories based on acoustic sources employed:

1. Airgun: seismic surveys and those HRG surveys that use airguns along with electromechanical equipment. These surveys are used under the Oil and Gas Program; and,
2. Non-airgun HRG: These surveys are used under the Renewable Energy and Marine Mineral Resource Programs and only involve electromechanical equipment.

These take estimates are presented in Table 11 for takes anticipated as a result of exposure to seismic airgun activities and Table 12 for takes anticipated as a result of exposure to non-airgun HRG surveys.

Blue Whales. The estimated number of blue whales that would be "taken" during the seismic and HRG survey activities BOEM plans to authorize off the Atlantic seaboard from 2013 through

2020 depends on the acoustic criteria being used. Using NMFS' 180 dB criterion for "level A harassment," resulted in a total of 15 instances in which a blue whale would be "taken" by all active sound sources associated with the G&G surveys over the 8-year period of the proposed action. All of the blue whales "taken" by this form of harassment would be "taken" as a result of their exposure to airguns associated with the surveys. However, using criteria for "level A harassment" proposed by Southall *et al.* (2007), 5 blue whales would be "taken" by exposure to sounds produced by the airgun surveys that BOEM proposes between 2013 and 2020.

Using NMFS' 160 dB criterion for "level B harassment," resulted in a total of 1,433 instances in which a blue whale would be "taken" by all active sound sources associated with the seismic and HRG surveys over the 8-year period of the proposed action. Based on the models BOEM used, all but 1 of these instances of "level B harassment" would result from exposure to airguns associated with the surveys; a single instance in which a blue whale would be "taken" by "level B harassment" by non-airgun HRG surveys over the 8-year period, although the probability of a blue whale being "taken" by this form of harassment in any particular year is less than 100%. That is, if we treat any estimate of the number of blue whales that might be "taken" that is smaller than 1.0 as a probability, the probability of blue whales being "taken" range from a low of 8.7% in 2019 to a high of 25.7% in 2020. In cases where a take estimate is non-zero and equal to or greater than 0.001, NMFS treats this as take of an individual animal since take of a portion of an individual is not possible.

Blue whales would primarily be exposed to the more offshore seismic and HRG surveys under the oil and gas program, not the more coastal surveys such as those under the renewable energy and marine mineral resource surveys.

Fin Whales. Like blue whales, the number of fin whales BOEM estimated would be "taken" during the seismic and HRG survey activities BOEM plans to authorize off the Atlantic seaboard from 2013 through 2020 depends on the acoustic criteria being used. Using NMFS' 180 dB criterion for "level A harassment," resulted in a total of 30 instances in which a fin whale would be "taken" by all active sound sources associated with the seismic and HRG surveys over the 8-year period of the proposed action (see Tables 11 and 12). All of the fin whales "taken" by this form of harassment would be "taken" as a result of their exposure to airguns. As with the blue whales, criteria for "level A harassment" proposed by Southall *et al.* (2007) produced different estimates: no fin whales would be "taken" by exposure to sounds produced by the seismic airgun surveys BOEM proposes between 2013 and 2020 (see Tables 11 and 12).

Using NMFS' 160 dB criterion for "level B harassment," resulted in a total of 3,025 instances in which a fin whale would be "taken" by all active sound sources associated with the seismic and HRG surveys over the 8-year period of the proposed action (see Tables 11 and 12). All but 1 of these instances of "level B harassment" would result from exposure to airguns; a single fin whale would be "taken" by "level B harassment" by non-airgun HRG surveys over the 8-year period, although the probability of a fin whale being "taken" by this form of harassment in any particular year is less than 100% (probabilities range from a low of 16.0% in 2013 to a high of 53.8% in 2020).

Fin whales would primarily be exposed in the mid-Atlantic Planning Area mostly in the fall, winter and spring.

Humpback Whales. The number of humpback whales BOEM estimated would be "taken" during the G&G activities BOEM plans to authorize off the Atlantic seaboard from 2013 through 2020 depends on the acoustic criteria being used. Using NMFS' 180 dB criterion for "level A harassment," resulted in a total of 38 instances in which a humpback whale would be "taken" by all active sound sources associated with the seismic and HRG survey activities over the 8-year period of the proposed action (see Tables 11 and 12). All of the humpback whales "taken" by this form of harassment would be "taken" as a result of their exposure to airguns associated with the survey activities. As with the other whales, criteria for "level A harassment" proposed by Southall *et al.* (2007) produced different estimates: 17 instances in which a humpback whale would be "taken" by exposure to sounds produced by the airgun surveys BOEM proposes between 2013 and 2020 (see Tables 11 and 12).

Using NMFS' 160 dB criterion for "level B harassment," resulted in a total of 3,829 humpback whales being "taken" by all active sound sources associated with the seismic and HRG surveys over the 8-year period of the proposed action (see Tables 11 and 12). All but 3 of these instances of "level B harassment" would result from exposure to airguns associated with the survey activities; a single humpback whale would be "taken" by "level B harassment" by non-airgun HRG surveys over the 8-year period, although the probability of a humpback whale being "taken" by this form of harassment in any particular year is less than 100% (probabilities range from a low of 21.9% in 2019 to a high of 64.9% in 2020).

Humpback whales would be exposed primarily during the winter months and in the mid-Atlantic Planning Area rather than the South Atlantic Planning Area.

North Atlantic Right Whales. As with all of the whales discussed thus far, the number of North Atlantic right whales BOEM estimated would be "taken" during the G&G activities BOEM plans to authorize off the Atlantic seaboard from 2013 through 2020 depends on the acoustic criteria being used. Using NMFS' 180 dB criterion for "level A harassment," resulted in a total of 9 instances in which a North Atlantic right whale would be "taken" by all active sound sources associated with the seismic and HRG surveys over the 8-year period of the proposed action (see Tables 11 and 12). All of the North Atlantic right whales "taken" by this form of harassment would be "taken" as a result of their exposure to airguns associated with the surveys. As with the other whales, criteria for "level A harassment" proposed by Southall *et al.* (2007) produced different estimates: no North Atlantic right whales would be "taken" by exposure to sounds produced by the airgun surveys BOEM proposes between 2013 and 2020 (see Tables 11 and 12).

Using NMFS' 160 dB criterion for "level B harassment," resulted in a total of 957 instances in which a North Atlantic right whale would be "taken" by all active sound sources associated with seismic and HRG surveys over the 8-year period of the proposed action (see Tables 11 and 12). All but 4 of these instances of "level B harassment" would result from exposure to airguns associated with the surveys; four North Atlantic right whales would be "taken" by "level B harassment" by non-airgun High Resolution Geophysical Surveys over the 8-year period, although the probability of a North Atlantic right whale being "taken" by this form of harassment in any particular year is less than 100% (probabilities range from a low of 19.5% in 2019 to a high of 87.0% in 2020).

Right whales would primarily be exposed to seismic and HRG survey activities in the mid-Atlantic Planning Area.

Sei Whales. As with the other whales discussed thus far, the number of sei whales BOEM estimated would be "taken" during the seismic and HRG survey activities BOEM plans to authorize off the Atlantic seaboard from 2013 through 2020 depends on the acoustic criteria being used. Using NMFS' 180 dB criterion for "level A harassment," resulted in a total of 14 sei whales being "taken" by all active sound sources associated with the seismic and HRG surveys over the 8-year period of the proposed action (see Tables 11 and 12). All of the sei whales "taken" by this form of harassment would be "taken" as a result of their exposure to airguns associated with the surveys. As with the other whales discussed thus far, criteria for "level A harassment" proposed by Southall *et al.* (2007) produced different estimates: no sei whales would be "taken" by exposure to sounds produced by airgun surveys BOEM proposes between 2013 and 2020 (see Tables 11 and 12).

Using NMFS' 160 dB criterion for "level B harassment," resulted in a total of 1,321 sei whales being "taken" by all active sound sources associated with the seismic and HRG surveys over the 8-year period of the proposed action (see Tables 11 and 12). .All but 1 of these instances of "level B harassment" would result from exposure to airguns associated with the surveys; four sei whales would be "taken" by "level B harassment" by non-airgun HRG surveys over the 8-year period, although the probability of a sei whale being "taken" by this form of harassment in any particular year is less than 100% (probabilities range from a low of 19.5% in 2019 to a high of 87.0% in 2020).

Sei whales would primarily be exposed in spring and fall possibly in both Planning Areas to seismic and HRG surveys.

Sperm Whales. As with the other whales discussed thus far, the number of sperm whales BOEM estimated would be "taken" during the seismic and HRG survey activities BOEM plans to authorize off the Atlantic seaboard from 2013 through 2020 depends on the acoustic criteria being used. Using NMFS' 180 dB criterion for "level A harassment," resulted in a total of 979 instances in which a sperm whale would be "taken" by all active sound sources associated with the seismic and HRG surveys over the 8-year period of the proposed action (see Tables 11 and 12). All of the sperm whales "taken" by this form of harassment would be "taken" as a result of their exposure to airguns associated with the surveys. As with the other whales discussed thus far, criteria for "level A harassment" proposed by Southall *et al.* (2007) produced different estimates: no sperm whales would be "taken" by exposure to sounds produced by airgun surveys BOEM proposes between 2013 and 2020 (see Tables 11 and 12).

Using NMFS' 160 dB criterion for "level B harassment," resulted in a total of 95,941 instances in which sperm whales would be "taken" by all active sound sources associated with the seismic and HRG surveys over the 8-year period of the proposed action (see Tables 11 and 12). All but 1 of these instances of "level B harassment" would result from exposure to airguns associated with the surveys; four sperm whales would be "taken" by "level B harassment" by non-airgun HRG surveys over the 8-year period, although the probability of a sperm whale being "taken" by this form of harassment is less than 100% in every year except for 2018, 2019, and 2020 (excluding these three years, probabilities range from a low of 22.0% in 2013, 2014, 2015 and 2016, to a high of 40.0% in 2017).

Sperm whales would be exposed primarily in the fall, winter and spring in the mid-Atlantic Planning Area.

Table 11: BOEM Take Estimates for Seismic and HRG Oil and Gas Surveys.

Species	BOEM Annual Estimates (2012 - 2020)									Total *	Annual Estimates based on BOEM Estimates (using t-distribution to compute confidence interval)			8-Year Estimate Based on Mean Annual Estimate and Confidence Intervals based on t-distribution		
	2012	2013	2014	2015	2016	2017	2018	2019	2020	2012-2020	Mean "Takes" Per Year	Lower 95% CI	Upper 95% CI	Total Takes Over 8 Years	Lower 95% CI	Upper 95% CI
Level A Harassment Takes (180-dB NMFS criterion)																
North Atlantic Right Whale	**0**	**1**	**2**	**0**	**2**	**1**	**1**	**1**	**1**	**9**	**1**	**0.42**	**1.58**	**8**	**4**	**13**
	0	1.162	2.291	0.269	1.883	1.394	1.294	0.874	0.595		1	0.48	1.69	9	4	14
Blue Whale	**0**	**2**	**4**	**0**	**3**	**2**	**2**	**1**	**1**	**15**	**2**	**0.59**	**2.75**	**14**	**5**	**22**
	0	2.182	4.274	0.451	2.753	2.001	1.540	0.882	0.537		2	0.53	2.72	13	5	22
Fin Whale	**0**	**4**	**9**	**1**	**6**	**4**	**3**	**2**	**1**	**30**	**3**	**1.03**	**5.64**	**27**	**9**	**46**
	0	4.399	8.637	0.949	5.896	4.342	3.475	1.924	1.223		3	1.19	5.66	28	10	46
Sei Whale	**0**	**2**	**4**	**0**	**3**	**2**	**1**	**1**	**1**	**14**	**2**	**0.47**	**2.64**	**13**	**4**	**22**
	0	1.966	3.855	0.417	2.565	1.880	1.476	0.818	0.502		1	0.50	2.49	12	5	20
Humpback Whale	**0**	**6**	**12**	**1**	**7**	**5**	**4**	**2**	**1**	**38**	**4**	**1.12**	**7.32**	**34**	**9**	**59**
	0.001	5.897	11.543	1.207	7.333	5.315	4.043	2.319	1.385		4	1.37	7.30	35	11	59
Sperm Whale	**0**	**159**	**310**	**30**	**180**	**127**	**89**	**55**	**30**	**980**	**109**	**29.45**	**188.33**	**872**	**236**	**1,507**

Species	**BOEM Annual Estimates (2012 - 2020)**									**Total * 2012-2020**	**Annual Estimates based on BOEM Estimates (using t-distribution to compute confidence interval)**			**8-Year Estimate Based on Mean Annual Estimate and Confidence Intervals based on t-distribution**		
	2012	**2013**	**2014**	**2015**	**2016**	**2017**	**2018**	**2019**	**2020**		**Mean "Takes" Per Year**	**Lower 95% CI**	**Upper 95% CI**	**Total Takes Over 8 Years**	**Lower 95% CI**	**Upper 95% CI**
	0.003	158.828	309.724	30.401	179.960	126.956	89.342	54.729	29.855		109	29.51	188.22	871	237	1,506
Level A Harassment Takes (Southall et al. 2007 criterion)																
North Atlantic Right Whale	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0.00**	**0.00**	**0**	**0**	**0**
	0	0.036	0.071	0.008	0.044	0.034	0.025	0.009	0.001		0	0.01	0.04	1	1	1
Blue Whale	**0**	**1**	**2**	**0**	**1**	**1**	**0**	**0**	**0**	**5**	**1**	**0.00**	**1.15**	**5**	**0**	**10**
	0	0.831	1.623	0.164	0.915	0.664	0.439	0.208	0.043		1	0.11	0.97	5	1	8
Fin Whale	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0.00**	**0.00**	**0**	**0**	**0**
	0	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000		0	0.00	0.00	**1**	**1**	**1**
Sei Whale	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0.00**	**0.00**	**0**	**0**	**0**
	0	0.019	0.403	0.033	0.176	0.113	0.057	0.060	0.030		0	0.00	0.20	1	0	2
Humpback Whale	**0**	**3**	**6**	**1**	**3**	**2**	**1**	**1**	**0**	**17**	**2**	**0.34**	**3.44**	**16**	**3**	**28**
	0	3.046	5.931	0.567	3.153	2.226	1.402	0.779	0.235		2	0.38	3.48	16	4	28
Sperm Whale	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0.00**	**0.00**	**0**	**0**	**0**
	0	0.095	0.184	0.015	0.076	0.050	0.021	0.019	0.001		0	0.00	0.10	1	1	1

Species	BOEM Annual Estimates (2012 - 2020)									Total *	Annual Estimates based on BOEM Estimates (using t-distribution to compute confidence interval)			8-Year Estimate Based on Mean Annual Estimate and Confidence Intervals based on t-distribution		
	2012	2013	2014	2015	2016	2017	2018	2019	2020	2012-2020	Mean "Takes" Per Year	Lower 95% CI	Upper 95% CI	Total Takes Over 8 Years	Lower 95% CI	Upper 95% CI
Level B Harassment Takes (160-dB NMFS criterion)																
North Atlantic Right Whale	**0**	**114**	**224**	**26**	**185**	**137**	**127**	**86**	**58**	**957**	**106**	**47.28**	**165.39**	**851**	**379**	**1,324**
	0	113.846	224.490	26.343	184.517	136.675	126.832	85.613	58.299		106	47.28	165.30	851	379	1,323
Blue Whale	**0**	**214**	**419**	**44**	**270**	**196**	**151**	**86**	**53**	**1,433**	**159**	**51.58**	**266.86**	**1,274**	**413**	**2,135**
	0	213.901	418.875	44.160	269.777	196.065	150.849	86.408	52.630		159	51.60	266.77	1,274	413	2,135
Fin Whale	**0**	**431**	**847**	**93**	**578**	**426**	**341**	**189**	**120**	**3,025**	**336**	**117.15**	**555.08**	**2,689**	**938**	**4,441**
	0	431.204	846.583	93.001	577.906	425.583	340.532	188.600	119.857		336	117.02	554.82	2,688	937	4,439
Sei Whale	**0**	**193**	**378**	**41**	**251**	**184**	**145**	**80**	**49**	**1,321**	**147**	**49.20**	**244.36**	**1,175**	**394**	**1,955**
	0	192.624	377.801	40.850	251.322	184.254	144.678	80.220	49.182		147	49.22	244.32	1,175	394	1,955
Humpback Whale	**0**	**578**	**1,131**	**118**	**719**	**521**	**396**	**227**	**136**	**3,826**	**425**	**134.57**	**715.65**	**3,401**	**1,077**	**5,726**
	0	577.965	1131.231	118.264	718.610	520.862	396.287	227.280	135.768		425	134.62	715.67	3,402	1,077	5,726
Sperm Whale	**0**	**15,567**	**30,356**	**2,980**	**17,549**	**12,443**	**8,756**	**5,364**	**2,926**	**95,941**	**10660**	**2889.35**	**18430.88**	**85,281**	**23,115**	**147,448**
	0	15566.706	30355.997	2979.612	17548.741	12442.986	8756.403	5363.975	2926.098		10660	2889.31	18430.80	85,281	23,115	147,447

*BOEM's estimated Annual Incidental Takes of Listed Marine Mammals Due to Seismic and HRG Oil and Gas Surveys. Source: Modified from Appendix E of the Programmatic EIS. NMFS's take estimates (in white) are the annual numbers of individuals taken, calculated to three decimal places and values were rounded to the nearest whole number of animals. The total is the sum of the rounded annual values. Mean values and confidence intervals were estimated based on the t-distribution and using the procedures and formulae described by Cumming (2012).

Table 12: BOEM Take Estimates for Non-Airgun HRG Surveys for Renewable Energy and Marine Minerals Program

Species	BOEM Annual Estimates (2012 - 2020)									Total *	Annual Estimates based on BOEM Estimates (using t-distribution to compute confidence interval)			8-Year Estimate Based on Mean Annual Estimate and Confidence Intervals based on t-distribution		
	2012	2013	2014	2015	2016	2017	2018	2019	2020	2012-2020	Mean "Takes" Per Year	Lower 95% CI	Upper 95% CI	Total Takes Over 9 Years	Lower 95% CI	Upper 95% CI
Level A Harassment Takes (180-dB NMFS criterion)																
North Atlantic Right Whale	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0.00**	**0.00**	**0**	**0**	**0**
	0.002	0.003	0.003	0.003	0.003	0.003	0.005	0.003	0.009		0	0.00	0.01	1	1	1
Blue Whale	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0.00**	**0.00**	**0**	**0**	**0**
	0.001	0.001	0.001	0.001	0.001	0.001	0.002	0.001	0.003		0	0.00	0.00	1	1	1
Fin Whale	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0.00**	**0.00**	**0**	**0**	**0**
	0.001	0.002	0.002	0.002	0.002	0.002	0.003	0.002	0.006		0	0.00	0.00	1	1	1
Sei Whale	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0.00**	**0.00**	**0**	**0**	**0**
	0.000	0.001	0.001	0.001	0.001	0.001	0.001	0.001	0.002		0	0.00	0.00	1	1	1
Humpback Whale	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0.00**	**0.00**	**0**	**0**	**0**
	0.003	0.003	0.004	0.004	0.004	0.003	0.004	0.002	0.007		0	0.00	0.00	1	1	1
Sperm Whale	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0.00**	**0.00**	**0**	**0**	**0**
	0.000	0.000	0.000	0.000	0.000	0.004	0.043	0.038	0.121		0	0.00	0.06	1	0	1

Species	BOEM Annual Estimates (2012 - 2020)									Total *	Annual Estimates based on BOEM Estimates (using t-distribution to compute confidence interval)			8-Year Estimate Based on Mean Annual Estimate and Confidence Intervals based on t-distribution		
	2012	2013	2014	2015	2016	2017	2018	2019	2020	2012-2020	Mean "Takes" Per Year	Lower 95% CI	Upper 95% CI	Total Takes Over 9 Years	Lower 95% CI	Upper 95% CI
Level A Harassment Takes (Southall et al. 2007 criterion)																
North Atlantic Right Whale	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0.00**	**0.00**	**0**	**0**	**0**
	0.002	0.003	0.003	0.003	0.003	0.002	0.001	0.000	0.000		0	0.00	0.00	1	1	1
Blue Whale	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**5**	**0**	**0.00**	**0.00**	**0**	**0**	**0**
	0.000	0.001	0.001	0.001	0.001	0.001	0.000	0.000	0.000		0	0.00	0.00	1	1	1
Fin Whale	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0.00**	**0.00**	**0**	**0**	**0**
	0.016	0.021	0.021	0.021	0.021	0.019	0.005	0.000	0.000		0	0.01	0.02	1	1	1
Sei Whale	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0.00**	**0.00**	**0**	**0**	**0**
	0.002	0.002	0.002	0.002	0.002	0.002	0.000	0.000	0.000		0	0.00	0.00	1	1	1
Humpback Whale	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**17**	**0**	**0.00**	**0.00**	**0**	**0**	**0**
	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000		0	0.00	0.00	0	0	0
Sperm Whale	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0.00**	**0.00**	**0**	**0**	**0**
	0.001	0.001	0.001	0.001	0.001	0.001	0.000	0.000	0.000		0	0.00	0.00	1	1	1
Level B Harassment Takes (160-dB NMFS criterion)																

Species	BOEM Annual Estimates (2012 - 2020)									Total *	Annual Estimates based on BOEM Estimates (using t-distribution to compute confidence interval)			8-Year Estimate Based on Mean Annual Estimate and Confidence Intervals based on t-distribution		
	2012	2013	2014	2015	2016	2017	2018	2019	2020	2012-2020	Mean "Takes" Per Year	Lower 95% CI	Upper 95% CI	Total Takes Over 9 Years	Lower 95% CI	Upper 95% CI
North Atlantic Right Whale	**0**	**0**	**0**	**0**	**0**	**0**	**1**	**0**	**1**	**2**	**0**	**0.00**	**0.58**	**2**	**0**	**5**
	0.195	0.246	0.249	0.249	0.249	0.269	0.502	0.300	0.870		0	0.17	0.52	3	2	5
Blue Whale	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0.00**	**0.00**	**0**	**0**	**0**
	0.066	0.098	0.104	0.104	0.104	0.105	0.145	0.087	0.247		0	0.07	0.16	1	1	2
Fin Whale	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**1**	**1**	**0**	**0.00**	**0.38**	**1**	**0**	**4**
	0.115	0.160	0.167	0.167	0.167	0.172	0.308	0.185	0.538		0	0.11	0.33	2	1	3
Sei Whale	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0.00**	**0.00**	**0**	**0**	**0**
	0.036	0.048	0.051	0.051	0.051	0.056	0.115	0.078	0.233		0	0.03	0.13	1	1	2
Humpback Whale	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**1**	**1**	**0**	**0.00**	**0.38**	**1**	**0**	**4**
	0.245	0.329	0.344	0.344	0.344	0.331	0.360	0.219	0.649		0	0.25	0.45	3	3	4
Sperm Whale	**0**	**0**	**0**	**0**	**0**	**0**	**4**	**4**	**12**	**20**	**2**	**0.00**	**5.53**	**18**	**0**	**45**
	0.018	0.022	0.022	0.022	0.022	0.405	4.213	3.697	11.891		2	0.00	5.51	19	0	45

*BOEM's estimated Annual Incidental Takes of Listed Marine Mammals Due to **Non-Airgun Surveys for Renewable Energy and Marine Minerals Programs**. Source: Modified from Appendix E of the Programmatic EIS. The total is the sum of the annual values, rounded to the nearest whole number of animals.NMFS's take estimates (in white) are the annual numbers of individuals taken, calculated to three decimal places and values were rounded to the nearest whole number of animals. Annual numbers of individuals taken were calculated to three decimal places, and values were rounded to the nearest whole number of animals. The total is the sum of the rounded annual values. Estimates over 0.001 in the Total Mean Takes Column are treated as 1 take. Mean values and confidence intervals were estimated based on the t-distribution and using the procedures and formulae described by Cumming (2012).

Exposure of Sea Turtles to G&G Activities

Neither BOEM nor NMFS was able to estimate takes from exposure to seismic and HRG surveys for sea turtles. As mentioned previously, estimated takes rely not only on the sound field as it propagates out from the source but also on virtual representations of the movements of individual animals within a sound field and the densities of those different species to be exposed to the sound field. Data upon which the models could be run for sea turtles is lacking, including behavioral parameters to run exposure models (i.e., dive patterns) and density estimates.

Without more specific information on the actual exposure of the endangered and threatened sea turtle species that we consider in this Opinion, we had to rely on the best scientific and commercial data available on sea turtles within the Action Area that indicates that green, hawksbill, Kemp's ridley, leatherback and Northwest Atlantic loggerheads can co-occur with G&G survey activities. Because we do not know the temporal or spatial aspects of individual G&G activities, we assume that sea turtles could be exposed to seismic and HRG surveys. Nesting, foraging and migrating turtles of all ages and life stages can occur within the Action Area.

In the western North Atlantic, green sea turtles can be found on various coastal beaches during the nesting season and at other times feeding or swimming along nearshore waters including embayments or other inshore habitats as well as offshore waters from Florida to Massachusetts (USDOC, NMFS and USDOI, USFWS, 2007a). Green turtles are also reported to use not only the coastal waters of North Carolina as summer foraging habitat, but the waters of Virginia as well (Mansfield et al., 2009). The Florida east coast is a prime foraging area for green turtles as well as possible sites in Virginia and south to Florida where macroalgae is found. Juvenile green turtles north of Florida must migrate south in autumn (Musick and Limpus, 1997). Therefore, neritic juvenile green turtles may occur within nearshore and inshore habitats throughout the Action Area. Green sea turtles could be exposed to seismic and HRG survey activities within the mid-Atlantic and south Atlantic Planning Areas during any time of the year.

Hawksbill sea turtles have stranded along almost the entire Atlantic coast of the United States, although most stranding records occur south of Cape Canaveral, Florida, particularly in Palm Beach, Broward and Miami-Dade counties (Florida Sea Turtle Stranding and Salvage database). Hawksbill sea turtles are very rare north of Florida, although they have been recorded as far north as Massachusetts. During their pelagic-stage, hawksbills disperse from the Gulf of Mexico and southern Florida in the Gulfstream Current, which could carry them offshore of Georgia and the Carolinas. There are records of hawksbill sea turtles stranding on the coast of Georgia (Ruckdeschel *et al.* 2000), being captured in pound nets off

Savannah, and being captured in summer flounder trawls (Epperly *et al.* 1995), gillnets (Epperly *et al.* 1995), and power plants off Georgia and the Carolinas.

Adult, female Kemp's ridleys rarely leave the Gulf of Mexico and adult males do not migrate. Juveniles feed along the east coast of the United States up to the waters off Cape Cod, Massachusetts (Spotila 2004). Juvenile Kemp's ridley sea turtles are the second most abundant sea turtle in the mid-Atlantic region from New England, New York, and the Chesapeake Bay, south to coastal areas off North Carolina (Morreale et al. 2007; TEWG 2000; Schmid 1998; Wibbels et al. 2005). The Mid Atlantic Bight is an important foraging area for juvenile Kemp's ridley turtles during spring through fall. Satellite tracking data document seasonal migration along the inner shelf of the eastern U.S. from New England to Florida. Wintering habitats for Kemp's ridley turtles in the northwestern Atlantic include shelf habitats off Florida and waters south of Cape Hatteras, North Carolina (Gitschlag, 1996). Kemp's ridley turtles display some seasonal and coastal migratory behavior; satellite tagging data indicate that Kemp's ridley turtles transit between nearshore and offshore waters (within 28 km [50 mi]) from spring/summer to fall/winter, which coincides with seasonal water temperature changes (USDOC, NMFS et al., 2010). The home ranges of Kemp's ridley turtles may be similar to those of loggerhead turtles (Shaver et al., 2005).

Leatherback turtles display wide ranging coastal and transoceanic movements (NMFS et al., 2010) and have the most wide-ranging distribution of any sea turtle. They regularly occur throughout the Action Area from coastal areas to deeper waters (>328 ft). Leatherbacks are reported from the North Sea, along the east coast of the U.S. and into the Gulf of Mexico and in the Caribbean Sea. Shrimp trawlers operating in the coastal waters from Cape Canaveral, Florida through North Carolina have interacted with leatherback sea turtles as they make their annual spring migration north. In April through June leatherbacks are found off South Carolina when cannonball jellyfish (*Stomolophus meleagris*) are abundant, and again in October and November during their fall migration (SCDNR, 2005b).

Northwest Atlantic Loggerheads

In U.S. Atlantic waters, loggerheads commonly occur from the beach to beyond the continental shelf from Florida to Cape Cod, Massachusetts, although their presence varies with the seasons due to changes in water temperature (Shoop and Kenney 1992; Mitchell *et al.* 2003; Braun-McNeill *et al.* 2008; Mansfield *et al.* 2009; Hawkes *et al.* 2006, 2011). Loggerhead sea turtles occur year round in ocean waters off North Carolina, South Carolina, Georgia, and Florida. As coastal water temperatures warm in the spring, loggerheads begin to migrate to inshore waters of the Southeast United States (*e.g.*, Pamlico and Core Sounds) and also move up the U.S. Atlantic Coast (Epperly *et al.* 1995a, 1995b, 1995c; Braun-McNeill and Epperly 2004), occurring in Virginia foraging areas as early as April/May and on the most northern foraging grounds in the Gulf of Maine in June (Shoop and Kenney 1992). The trend is reversed in the fall as water temperatures cool. The large majority leave the Gulf of Maine by mid-September but some turtles may remain in Mid-Atlantic and Northeast areas until late fall. By December, loggerheads have migrated from inshore and more northern coastal waters to waters offshore of North Carolina, particularly off of Cape Hatteras, and waters further south where the influence of the Gulf Stream provides temperatures favorable to sea turtles (Shoop and Kenney 1992; Epperly *et al.* 1995b).

Atlantic Sturgeon

Neither BOEM nor NMFS was able to estimate takes for Atlantic sturgeon. As mentioned previously, estimated takes rely not only on the sound field as it propagates out from the source but also on virtual representations of the movements of individual animals within a sound field and the densities of those different species to be exposed to the sound field. Data upon which the models could be run for sea turtles is lacking, including behavioral parameters (i.e., dive patterns) and density estimates.

Without more specific information on the actual exposure of the endangered on Atlantic sturgeon, we had to rely on the best scientific and commercial data available on Atlantic sturgeon within the Action Area that indicates that these fish can co-occur with G&G survey activities, primarily HRG surveys.

Atlantic sturgeon remain in the natal estuary for months to years before emigrating to open ocean as subadults (Holland and Yelverton 1973; Dovel and Berggen 1983; Waldman et al. 1996; Dadswell 2006; ASSRT 2007). Subadults and adults travel within the marine environment, where they may undergo extensive movements usually confined to shelly or gravelly bottoms in 10-50 m (33-164 ft) water depths (Stein et al., 2004; Erickson et al., 2011).

Evidence of seasonal movements are apparent from tag returns: Atlantic sturgeon originating from the Delaware River estuary were recapture during the fall in nearshore waters along the Atlantic coast as far south as Cape Hatteras, North Carolina from November through early March. In the spring, a portion of the tagged fish reentered the Delaware River estuary, however, many fish continued a northerly coastal migration into southern New England waters where they were recovered throughout the summer months. Movements as far north as Maine were documented. A southerly coastal migration was apparent from tag returns reported in the fall. The majority of these tag returns were reported from relatively shallow near shore fisheries with few fish reported from waters in excess of 25 m (C. Shirey, Delaware Department of Fish and Wildlife, unpublished data reviewed in ASMFC, 2009). Areas where migratory Atlantic sturgeon commonly aggregate include Delaware Bay, Chesapeake Bay, and from the Virginia/North

Carolina border to Cape Hatteras at depths up to 24 m. Based on this information about migrating Atlantic sturgeon, individuals originating from all of the 5 listed DPSs may occur in the Action Area.

Fish distribution varies seasonally within this depth range. During summer months (May to September) fish are primarily found in the shallower depths of 10-20 m (33-66 ft). In winter and early spring (December to March), fish move to depths between 20 and 50 m (66 and 165 ft) (Erickson et al., 2011). Shelf areas <18 m (59 ft) deep off Virginia and the sandy shoals offshore of Oregon Inlet, North Carolina, appear to be areas of concentration during summer months (Laney et al., 2007). The area of high concentration offshore of Virginia was centered from 15 to 37.5 km (9.3 to 23.3 mi) from shore, and the maximum distance from shore during winter was about 112.5 km (70 mi).

Response Analyses

As discussed in the *Approach to the Assessment* section of this biological opinion, response analyses determine how listed resources are likely to respond after being exposed to an Action's effects on the environment or directly on listed species themselves. For the purposes of consultations on seismic and HRG surveys, our assessments try to detect the probability of lethal responses, sensory impairment (permanent and temporary threshold shifts and acoustic masking), physiological responses (particular stress responses), behavioral responses, and social responses that might result in reducing the fitness of listed individuals. Ideally, our response analyses consider and weigh evidence of adverse consequences, beneficial consequences, or the absence of such consequences.

Without more specific empirical information on the actual responses of the endangered and threatened species that we consider in this Opinion upon being exposed to sounds produced by equipment employed during seismic surveys, we had to rely on the best scientific and commercial data available to assess the probable responses of endangered and threatened species to these sounds. In the narratives that follow this introduction, we summarize the best scientific and commercial data on the responses of marine animals to sounds produced by equipment employed during seismic and HRG surveys and other acoustic stimuli. Then we use that information to make inferences about the probable responses of the endangered and threatened species we are considering in this Opinion.

Potential Responses of Listed Species to Vessel Traffic

Numerous studies of interactions between vessels and marine mammals have demonstrated that free-ranging marine mammals engage in avoidance behavior when vessels move toward them. It is not clear whether these responses are caused by the physical presence of a surface vessel, the underwater noise generated by the vessel, or an interaction between the two (Goodwin and Green 2004; Lusseau 2006). However, several authors suggest that the noise generated during motion is probably an important factor (Blane and Jackson 1994, Evans *et al.* 1992, 1994). These studies suggest that marine mammal behavioral responses to vessels are similar to their behavioral responses to predators.

As we discussed previously, based on the suite of studies of cetacean behavior to vessel approaches (Au and Green 1990, Au and Perryman 1982, Bain *et al.* 2006, Bauer 1986, Bejder 1999, 2006a, 2006b; Bryant *et al.* 1984, Corkeron 1995, David 2002, Erbé 2000, Félix 2001, Magalhães *et al.* 2002, Goodwin and Cotton 2004, Hewitt 1985, Lusseau 2003, 2006; Lusseau and Bejder 2007, Ng and Leung 2003, Nowacek *et al.* 2001, Richter *et al.* 2003, 2006; Scheidat *et al.* 2004, Simmonds 2005, Watkins 1986, Williams and Ashe 2007, Williams *et al.* 2002, 2006a, 2006b; Würsig *et al.* 1998), the set of variables that help determine whether marine mammals are likely to be disturbed by vessels include:

1. *number of vessels*. The behavioral repertoire marine mammals have used to avoid interactions with surface vessels appears to depend on the number of vessels in their perceptual field (the area within which animals detect acoustic, visual, or other cues) and the animal's assessment of the risks associated with those vessels (the primary index of risk is probably vessel proximity relative to the animal's flight initiation distance).
   Below a threshold number of vessels (which probably varies from one species to another, although groups of marine mammals probably share sets of patterns), studies have shown that whales will attempt to avoid an interaction using horizontal avoidance behavior[3]. Above that threshold, studies have shown that marine mammals will tend to avoid interactions using vertical avoidance behavior, although some marine mammals will combine horizontal avoidance behavior with vertical avoidance behavior (Bryant *et al.* 1984, Cope *et al.* 2000, David 2002, Lusseau 2003, Kruse 1991, Nowacek *et al.* 2001, Stensland and Berggren 2007, Williams and Ashe 2007);
2. *the distance between vessel and marine mammals* when the animal perceives that an approach has started and during the course of the interaction (Au and Perryman 1982, David 2002, Hewitt 1985, Kruse 1991);
3. *the vessel's speed and vector* (David 2002);
4. *the predictability of the vessel's path*. That is, cetaceans are more likely to respond to approaching vessels when vessels stay on a single or predictable path (Acevedo 1991, Angradi *et*

---

[3] As discussed in the *Approach to the Assessment* section of this Opinion, we distinguish between "avoidance," "evasion," and "escape" using the distinctions proposed by Weihs and Webb (1984): "avoidance" is a shift in position by prey before a potential predator begins an attack; "evasion" is a response by potential prey to a perceived attack from a potential predator; and "escape" is the most acute form of evasive behavior.

al. 1993; Browning and Harland 1999; Lusseau 2003, 2006; Williams *et al.* 2002, 2006a, 2006b) than when it engages in frequent course changes (Evans *et al. 1*994, Lusseau 2006, Williams *et al.* 2002)

5. *noise associated with the vessel* (particularly engine noise) and the rate at which the engine noise increases (which the animal may treat as evidence of the vessel's speed; David 2002, Lusseau 2003, 2006);
6. *the type of vessel* (displacement versus planing), which marine mammals may interpret as evidence of a vessel's maneuverability (Goodwin and Cotton 2004);
7. the behavioral state of the marine mammals (David 2002, Lusseau 2003, 2006; Würsig *et al.* 1998). For example, Würsig *et al.* (1998) concluded that whales were more likely to engage in avoidance responses when the whales were "milling" or "resting" than during other behavioral states.

Most of the investigations cited earlier reported that animals tended to reduce their visibility at the water's surface and move horizontally away from the source of disturbance or adopt erratic swimming strategies (Corkeron 1995, Lusseau 2003, Lusseau 2004, 2005*a*; Notarbartolo di Sciara *et al.* 1996, Nowacek *et al.* 2001, Van Parijs and Corkeron 2001, Williams *et al.* 2002). In the process, their dive times increased, vocalizations and jumping were reduced (with the exception of beaked whales), individuals in groups move closer together, swimming speeds increased, and their direction of travel took them away from the source of disturbance (Edds and Macfarlane 1987, Baker and Herman 1989, Kruse 1991, Polacheck and Thorpe 1990, Evans *et al. 1*992, Lütkebohle 1996, Nowacek *et al. 1*999). Some individuals also dove and remained motionless, waiting until the vessel moved past their location. Most animals finding themselves in confined spaces, such as shallow bays, during vessel approaches tended to move towards more open, deeper waters (Stewart *et al.* 1982, Kruse 1991). We assume that this movement would give them greater opportunities to avoid or evade vessels as conditions warranted.

Although most of these studies focused on small cetaceans (for example, bottlenose dolphins, spinner dolphins, spotted dolphins, harbor porpoises, beluga whales, and killer whales), studies of large whales have reported similar results for fin and sperm whales (David 2002, Notarbartolo di Sciara *et al.* 1996, 2002). Baker *et al.* (1983) reported that humpbacks in Hawai'i responded to vessels at distances of 2 to 4 km. Richardson *et al.* (1985) reported that bowhead whales (*Balaena mysticetus*) swam in the opposite direction of approaching seismic vessels at distances between 1 and 4 km and engage in evasive behavior at distances under 1 km. Fin whales also responded to vessels at a distances of about 1 km (Edds and Macfarlane 1987).

Some cetaceans detect the approach of vessels at substantial distances. Finley *et al.* (1990) reported that beluga whales seemed aware of approaching vessels at distances of 85 km and began to avoid the approach at distances of 45-60 km. Au and Perryman (1982) studied the behavioral responses of eight schools of spotted and spinner dolphins (*Stenella attenuata* and *S. longirostris*) to an approaching ship (the NOAA vessel *Surveyor*: 91.4 meters, steam-powered, moving at speeds between 11 and 13 knots) in the eastern Pacific Ocean (10°15 N lat., 109°10 W long.). They monitored the response of the dolphin schools to the vessel from a Bell 204 helicopter flying a track line ahead of the ship at an altitude of 366 – 549 meters (they also monitored the effect of the helicopter on dolphin movements and concluded that it had no observable effect on the behavior of the dolphin schools). All of the schools continuously adjusted their direction of swimming by small increments to continuously increase the distance between the school and the ship over time. The animals in the eight schools began to flee from the ship at distances ranging from 0.9 to 6.9 nm. When the ship turned toward a school, the individuals in the school increased their swimming speeds (for example, from 2.8 to 8.4 knots) and engaged in sharp changes in direction.

Hewitt (1985) reported that five of 15 schools of dolphin responded to the approach of one of two ships used in his study and none of four schools of dolphin responded to the approach of the second ship (the first ship was the NOAA vessel *David Jordan Starr*; the second ship was the *Surveyor*). Spotted dolphin and spinner dolphins responded at distances between 0.5 to 2.5 nm and maintained distances of 0.5 to 2.0 nm from the ship while striped dolphins allowed much closer approaches. Lemon *et al.*(2006) reported that bottlenose dolphins began to avoid approaching vessels at distances of about 100 m.

Würsig *et al.* (1998) studied the behavior of cetaceans in the northern Gulf of Mexico in response to survey vessels and aircraft. They reported that *Kogia* species and beaked whales (ziphiids) showed the strongest avoidance reactions to approaching ships (avoidance reactions in 11 of 13 approaches) while spinner dolphins, Atlantic spotted dolphins, bottlenose dolphins, false killer whales, and killer whales either did not respond or approached the ship (most commonly to ride the bow). Four of 15 sperm whales avoided the ship while the remainder appeared to ignore its approach.

Because of the number of vessels involved in seismic and HRG surveys, their slow speeds, the predictability of their track lines (which involve limited course changes), and sounds associated with their engines and displacement of water along their bowline, the available evidence leads us to expect marine mammals to treat survey vessels as potential stressors. Animals that perceive an approaching potential predator, predatory stimulus, or disturbance stimulus have four behavioral options (see Blumstein 2003 and Nonacs and Dill 1990):

a. ignore the disturbance stimulus entirely and continue behaving as if a risk of predation did not exist;
b. alter their behavior in ways that minimize their perceived risk of predation, which generally involves fleeing immediately;
c. change their behavior proportional to increases in their perceived risk of predation which requires them to monitor the behavior of the predator or predatory stimulus while they continue their current activity, or
d. take proportionally greater risks of predation in situations in which they perceive a high gain and proportionally lower risks where gain is lower, which also requires them to monitor the behavior of the predator or disturbance stimulus while they continue their current activity.

The latter two options are energetically costly and reduce benefits associated with the animal's current behavioral state. As a result, animals that detect a predator or predatory stimulus at a greater distance are more likely to flee at a greater distance (see Holmes *et al.* 1993, Lord *et al.* 2001). Some investigators have argued that short-term avoidance reactions can lead to longer term impacts such as causing marine mammals to avoid an area (Salden 1988, Lusseau 2005) or alter a population's behavioral budget (Lusseau 2004) which could have biologically significant consequences on the energetic budget and reproductive output of individuals and their populations.

Based on knowledge of the sensory biology of sea turtles (Bartol and Musick, 2003; Levenson *et al*., 2004; Ketten and Bartol, 2006; Moein Bartol and Ketten, 2006), they are likely to detect approaching water vessels via auditory or visual cues, however, there is limited information on how sea turtles are likely to respond to vessel approaches. Hazel *et al*. (2007) reported that sea turtles reacted to approaching vessels in a variety of ways. Benthic turtles launched upwards at a shallow angle and began swimming. The majority of the turtles swam away from the vessel while some swam along the vessel's track and some crossed in front of the vessel's track before swimming away. Sea turtle reaction time was greatly dependent on the speed of the vessel; sea turtles were able to react faster to slower moving vessels than to faster moving vessels. All of these responses were short-term responses that did not seem likely to have adverse long-term consequences for the individual sea turtles. Hazel *et al*. (2007) suggested that green sea turtles rely more on visual cues than auditory cues when reacting to approaching water vessels. Of the endangered and threatened species that occur in the Action Area for this consultation, the endangered and threatened sea turtles are most likely to ignore seismic survey vessels entirely and continue behaving as if the vessels and any risks associated with those vessels did not exist.

Vessel noise produces sounds in the general hearing range of fishes (Amoser et al., 2004). Additionally, vessels (i.e., trawlers, ferries, small boats) can change fish behavior (e.g., induce avoidance, alter swimming speed and direction, and alter schooling behavior) (Sarà et al. 2007). The sounds produced by motor-driven ships causes herring to dive and swim away from the vessel (Mitson and Knudsen, 2003). Sand et al. (2008) have pointed out that passing ships produce high levels of infrasonic and low-frequency noise (>10-1,000 Hz), and that infrasonic frequencies may be responsible for the observed avoidance reactions.

Review of Literature on the Potential Responses of Listed Species to Acoustic Stimuli

As discussed in the *Approach to the Assessment* section of this Opinion, we conduct response analyses to determine whether and how listed species and designated critical habitat are likely to respond after being exposed to an Action's effects. For the purposes of consultations on activities that involve seismic surveys, our assessments try to detect the probability of lethal responses, sensory impairment (permanent and temporary threshold shifts and acoustic masking), physiological responses (particular stress responses), behavioral responses, and social responses that are likely to directly or indirectly reduce the fitness of listed individuals.

Our response analyses consider and weigh all of the evidence available on the response of marine animals upon being exposed to seismic and electromechanical survey activities and the probable fitness consequences for the animals that exhibit particular responses or sequence of responses. It is important to acknowledge, however, that the empirical evidence on how endangered or threatened marine animals respond upon being exposed to sounds produced by equipment employed during seismic surveys in natural settings is very limited. Therefore, the narratives that follow this introduction summarize the best scientific and commercial data available on the responses of other species to sounds produced by equipment employed during seismic and HRG surveys. Based on those data, we identify the probable responses of endangered and threatened marine animals to seismic and HRG transmissions.

Figure 3 illustrates the conceptual model we use to assess the potential responses of marine animals when they are exposed to sounds produced by equipment employed during seismic surveys. The narratives that follow are generally organized around the items listed in the column titled "Proximate Responses by Category" in that Figure. These analyses examine the evidence available to determine if exposing endangered and threatened species to sounds produced by equipment employed during seismic surveys is likely to cause responses that might reduce the fitness of individuals that might be exposed.

Physical Damage

For the purposes of this assessment, “injuries” represents physical trauma or damage that is a direct result of an acoustic exposure, regardless of the potential consequences of those injuries to an animal (we distinguish between injuries that result from an acoustic exposure and injuries that result from an animal’s behavioral reaction to an acoustic exposure, which is discussed later in this section of the Opinion). Based on the literature available, sounds produced by equipment employed during seismic surveys might injure marine animals through two mechanisms (see “Box P” in Figure 3): acoustic resonance and noise-induced loss of hearing sensitivity (more commonly-called “threshold shift”).

ACOUSTIC RESONANCE. Acoustic resonance results from hydraulic damage in tissues that are filled with gas or air that resonates when exposed to acoustic signals (Box P1 of Figure 2 illustrates the potential consequences of acoustic resonance; see Rommel *et al*. 2007). Based on studies of lesions in beaked whales that stranded in the Canary Islands and Bahamas associated with exposure to naval exercises that involved sonar, investigators have identified two physiological mechanisms that might explain some of those stranding events: tissue damage resulting from resonance effects (Ketten 2004, Cudahy and Ellison 2001) and tissue damage resulting from “gas and fat embolic syndrome” (Fernandez *et al*. 2005, Jepson *et al*. 2003, 2005). Fat and gas embolisms are believed to occur when tissues are supersaturated with dissolved nitrogen gas and diffusion facilitated by bubble-growth is stimulated within those tissues (the bubble growth results in embolisms analogous to the “bends” in human divers).

Cudahy and Ellison (2001) analyzed the potential for resonance from low frequency sonar signals to cause injury and concluded that the expected threshold for *in vivo* (in the living body) tissue damage for underwater sound is on the order of 180 to 190 dB. There is limited direct empirical evidence (beyond Schlundt *et al*. 2000) to support a conclusion that 180 dB is “safe” for marine mammals; however, evidence from marine mammal vocalizations suggests that 180 dB is not likely to physically injure marine mammals. For example, Frankel (1994) estimated the source level for singing humpback whales to be between 170 and 175 dB; McDonald *et al*. (2001) calculated the average source level for blue whale calls as 186 dB, Watkins *et al*. (1987) found source levels for fin whales up to 186 dB, and Møhl *et al*. (2000) recorded source levels for sperm whale clicks up to 223 dB_{rms}. Because whales are not likely to communicate at source levels that would damage the tissues of other members of their species, this evidence suggests that these source levels are not likely to damage the tissues of the endangered and threatened species being considered in this consultation.

Crum and Mao (1996) hypothesized that received levels would have to exceed 190 dB in order for there to be the possibility of significant bubble growth due to super-saturation of gases in the blood. Jepson *et al*. (2003, 2005) and Fernández *et al*. (2004, 2005) concluded that *in vivo* bubble formation, which may be exacerbated by deep, long-duration, repetitive dives may explain why beaked whales appear to be particularly vulnerable to exposure to mid-frequency active sonar (as opposed to low-frequency seismic sound sources).

Based on the information available, the endangered or threatened marine mammals and sea turtles that we are considering in this Opinion are not likely to experience acoustic resonance. All of the evidence available suggests that this phenomenon poses potential risks to smaller cetaceans like beaked whales rather than the larger cetaceans that have been listed as endangered. Thus far, this phenomenon has not been reported for or associated with sea turtles, perhaps because they do not engage in dive patterns that are similar to those of beaked whales.

NOISE-INDUCED LOSS OF HEARING SENSITIVITY. Noise-induced loss of hearing sensitivity[4] or "threshold shift" refers to an ear's reduced sensitivity to sound following exposure to loud noises; when an ear's sensitivity to sound has been reduced, sounds must be louder for an animal to detect and recognize it. Noise-induced loss of hearing sensitivity is usually represented by the increase in intensity (in decibels) sounds must have to be detected. These losses in hearing sensitivity rarely affect the entire frequency range an ear might be capable of detecting, instead, they affect the frequency ranges that are roughly equivalent to or slightly higher than the frequency range of the noise itself. Nevertheless, most investigators who study TTS in marine mammals report the frequency range of the "noise," which would change as the spectral qualities of a waveform change as it moves through water, rather than the frequency range of the animals they study. Without information on the frequencies of the sounds we consider in this Opinion at the point at which it is received by endangered and threatened marine mammals, we assume that the frequencies are roughly equivalent to the frequencies of the source.

Acoustic exposures can result in three main forms of noise-induced losses in hearing sensitivity: permanent threshold shift, temporary threshold shift, and compound threshold shift (Miller 1974, Ward 1998, Yost 2007). When permanent loss of hearing sensitivity, or PTS, occurs, there is physical damage to the sound receptors (hair cells) in the ear that can result in total or partial deafness, or an animal's hearing

---

[4] Animals can experience losses in hearing sensitivity through other mechanisms. The processes of aging and several diseases cause some humans to experience permanent losses in their hearing sensitivity. Body burdens of toxic chemicals can also cause animals, including humans, to experience permanent and temporary losses in their hearing sensitivity (for example, see Mills and Going 1982 and Fechter and Pouyanos 2005).

can be permanently impaired in specific frequency ranges, which can cause the animal to be less sensitive to sounds in that frequency range. Traditionally, investigations of temporary loss of hearing sensitivity, or TTS, have focused on sound receptors (hair cell damage) and have concluded that this form of threshold shift is temporary. Hair cell damage does not accompany TTS and loss in hearing sensitivity are short-term and are followed by a period of recovery to pre-exposure hearing sensitivity that can last for minutes, days, or weeks. More recently, however, Kujawa and Liberman (2009) reported on noise-induced degeneration of the cochlear nerve that is a delayed result of acoustic exposures that produce TTS, that occurs in the absence of hair cell damage, and that is irreversible. They concluded that the reversibility of noise-induced threshold shifts, or TTS, can disguise progressive neuropathology that would have long-term consequences on an animal's ability to process acoustic information. If this phenomenon occurs in a wide range of species, TTS may have more permanent effects on an animal's hearing sensitivity than earlier studies would lead us to recognize.

Compound threshold shift or CTS, occurs when some loss in hearing sensitivity is permanent and some is temporary (for example, there might be a permanent loss of hearing sensitivity at some frequencies and a temporary loss at other frequencies or a loss of hearing sensitivity followed by partial recovery).

Although the published body of science literature contains numerous theoretical studies and discussion papers on hearing impairments that can occur with exposure to a strong sound, only a few studies provide empirical information on noise-induced loss in hearing sensitivity in marine mammals. Most of the few studies available have reported the responses of captive animals exposed to sounds in controlled experiments. Schlundt *et al.* (2000; see also Finneran *et al.* 2001, 2003) provided a detailed summary of the behavioral responses of trained marine mammals during TTS tests conducted at the Navy's SPAWAR Systems Center with 1-second tones. Schlundt *et al.* (2000) reported on eight individual TTS experiments that were conducted in San Diego Bay. Fatiguing stimuli durations were 1 second. Because of the variable ambient noise in the bay, low-level broadband masking noise was used to keep hearing thresholds consistent despite fluctuations in the ambient noise.

Finneran *et al.* (2001, 2003) conducted TTS experiments using 1-second duration tones at 3 kHz. The test method was similar to that of Schlundt *et al.* except the tests were conducted in a pool with a very low ambient noise level (below 50 dB re 1 $\mu Pa^2/Hz$), and no masking noise was used. The signal was a sinusoidal amplitude modulated tone with a carrier frequency of 12 kHz, modulating frequency of 7 Hz, and SPL of approximately 100 dB re 1 μPa. Two separate experiments were conducted. In the first, fatiguing sound levels were increased from 160 to 201 dB SPL. In the second experiment, fatiguing sound levels between 180 and 200 dB re 1 μPa were randomly presented.

There is no empirical evidence that exposure to seismic or sonar transmissions with this kind of intensity can cause PTS in any marine mammals; instead the probability of PTS has been inferred from studies of TTS (see Richardson *et al* 1995). However, Kujawa and Liberman (2009) argued that traditional testing of threshold shifts, which have focused on recovery of threshold sensitivities after exposure to noise, would miss acute loss of afferent nerve terminals and chronic degeneration of the cochlear nerve, which would have the effect of permanently reducing an animal's ability to perceive and process acoustic signals. Based on their studies of small mammals, Kujawa and Liberman (2009) reported that two hours of acoustic exposures produced moderate temporary threshold shifts but caused delayed losses of afferent nerve terminals and chronic degeneration of the cochlear nerve in test animals.

Although sea turtles detect low frequency sound, the potential effects on sea turtle biology remain largely unknown (Samuel, 2005). TTS in loggerhead sea turtles is reported to have been observed during studies by Moein *et al.* (1994). Turtle hearing was tested before, within 24 hours after, and two weeks after exposure to pulses of airgun sound. Moein *et al.* (1994) used an evoked potential method to test sea turtle hearing and concluded that the turtles exhibited some change in their hearing when tested within 24 hours after exposure (relative to pre-exposure hearing). The authors found that hearing had reverted to normal when tested two weeks after exposure. The size of the airgun used or the received sounds levels were not provided; therefore, the levels of airgun sounds that apparently elicited TTS are not known. These findings indicate that TTS may occur in sea turtles exposed to seismic sources; however, turtles in the study were confined. Sea turtles at sea might exhibit avoidance behavior and, thus, reduced exposure to seismic pulses. For any sea turtles that exhibit little or no behavioural avoidance, or if turtles habituate to seismic noise such that avoidance reactions cease, these individuals could sustain hearing loss if exposed to high enough sound levels from seismic airguns.

The sensory cells responsible for hearing in the mammalian ear and the fish ear are the same. The difference between fishes and mammals is that fishes retain a regenerative mechanism in the ear for when cells are lost, whereas no such capacity is found in mammals (BOEM 2012a). Popper and Hastings (2009b) reviewed data on TTS in fishes which suggest that TTS occurs after long-term exposure to sounds that are as high as 170-180 dB re 1 μPa (rms), but only in species that have specializations that result in their having relatively wide hearing bandwidths (to over 2 kHz) and lower hearing thresholds than fishes without specializations (which Atlantic sturgeon do not). If TTS occurs as a result of exposure to loud sounds, it is not necessarily very great and recovery seems to be within 24 hrs in most cases (Popper et al., 2005, 2007; Hastings et al., 2008; Hastings and Miskis-Olds, 2011). However, if the hearing ability of an affected fish decreases, then the likelihood of detecting predators, prey, or mates decline, thus decreasing the potential fitness of the receiver until normal hearing returns (BOEM 2012a).

Several variables affect the amount of loss in hearing sensitivity: the level, duration, spectral content, and temporal pattern of exposure to an acoustic stimulus as well as differences in the sensitivity of individuals and species. All of these factors combine to determine whether an individual organism is likely to experience a loss in hearing sensitivity as a result of acoustic exposure (Miller 1974, Ward 1998, Yost 2007). In free-ranging species, an animal's behavioral responses to a single acoustic exposure or a series of acoustic exposure events would also determine whether the animal is likely to experience losses in hearing sensitivity as a result of acoustic exposure. Unlike humans whose occupations or living conditions expose them to sources of potentially-harmful noise, in most circumstances, free-ranging animals are not likely to remain in a sound field that contains potentially harmful levels of noise unless they have a compelling reason to do so (for example, if they must feed or reproduce in a specific location). Any behavioral responses that would take an animal out of a sound field entirely or reduce the intensity of an exposure would reduce the animal's probability of experiencing noise-induced losses in hearing sensitivity.

More importantly, the data on captive animals and the limited information from free-ranging animals suggests that temporary noise-induced hearing losses do not have direct or indirect effect on the longevity or reproductive success of animals that experience permanent, temporary, or compound threshold shifts (Box P2 of Figure 2 illustrates the potential consequences of noise-induced loss in hearing sensitivity). Like humans, free-ranging animals might experience short-term impairment in their ability to use their sense of hearing to detect environmental cues about their environment while their ears recover from the temporary loss of hearing sensitivity. Although we could not locate information about how animals that experience noise-induced hearing loss alter their behavior or the consequences of any altered behavior on the lifetime reproductive success of those individuals, the limited information available would not lead us to expect temporary losses in hearing sensitivity to incrementally reduce the lifetime reproductive success of animals.

Behavioral Responses

Marine mammals, sea turtles and fish have not had the time and have not experienced the selective pressure necessary for them to have evolved a behavioral repertoire containing a set of potential responses to sounds produced by equipment employed during seismic surveys or human disturbance generally. Instead, marine animals invoke behavioral responses that are already in their behavioral repertoire to decide how they will behaviorally respond to airguns or electromechanical equipment, other potential stressors associated with seismic or HRG surveys or human disturbance generally. An extensive number of studies have established that these animals will invoke the same behavioral responses they would invoke when faced with predation and will make the same ecological considerations when they experience human disturbance that they make when they perceive they have some risk of predation (Beale and Monaghan 2004, Bejder *et al*. 2009, Berger *et al*. 1983, Frid 2003, Frid and Dill 2002, Gill *et al*. 2000, 2001; Gill and Sutherland 2000, 2001; Harrington and Veitch 1992, Lima 1998, Lima & Dill 1990, Madsen 1994, Romero 2004). Specifically, when animals are faced with a predator or predatory stimulus, they consider the risks of predation, the costs of anti-predator behavior, and the benefits of continuing a pre-existing behavioral pattern when deciding which behavioral response is appropriate in a given circumstance (Bejder *et al*. 2009, Gill *et al*. 2001, (Houston and McNamara 1986, Lima 1998, Lima and Bednekoff 1999, Ydenberg and Dill 1996). Further, animals appear to detect and adjust their responses to temporal variation in predation risks (Kat and Dill 1998, Lima and Bednekoff 1999, Rodriguez-Prieto *et al*. 2008).

The level of risk an animal perceives results from a combination of factors that include the perceived distance between an animal and a potential predator, whether the potential predator is approaching the animal or moving tangential to the animal, the number of times the potential predator changes its vector (or evidence that the potential predator might begin an approach), the speed of any approach, the availability of refugia, and the health or somatic condition of the animal, for example, along with factors related to natural predation risk (e.g., Frid 2001, Frid and Dill 2002, Papouchis *et al*. 2001). In response to a perceived threat, animals can experience physiological changes that prepare them for flight or fight responses or they can experience physiological changes with chronic exposure to stressors that have more serious consequences such as interruptions of essential behavioral or physiological events, alteration of an animal’s time budget, or some combinations of these responses (Frid and Dill 2002, Romero 2004, Sapolsky *et al*. 2000, Walker *et al*. 2005).

The behavioral responses of animals to human disturbance have been documented to cause animals to abandon nesting and foraging sites (Bejder *et al*. 2009, Gill *et al*. 2001, Sutherland and Crockford 1993), cause animals to increase their activity levels and suffer premature deaths or reduced reproductive success when their energy expenditures exceed their energy budgets (Daan *et al*. 1996, Feare 1976, Giese 1996, Mullner *et al*. 2004, Waunters *et al*. 1997), or cause animals to experience higher predation rates when they adopt risk-prone foraging or migratory strategies (Frid and Dill 2002).

Based on the evidence available from empirical studies of animal responses to human disturbance, marine animals are likely to exhibit one of several behavioral responses upon being exposed to sounds produced by equipment employed during seismic surveys: (1) they may engage in horizontal or vertical avoidance behavior to avoid exposure or continued exposure to a sound that is painful, noxious, or that they perceive as threatening (Boxes BR1.1 and BR1.2 of Figure 2); (2) they may engage in evasive behavior to escape exposure or continued exposure to a sound that is painful, noxious, or that they perceive as threatening, which we would assume would be accompanied by acute stress physiology (Box BR1.3 of Figure 2); (3) they may remain continuously vigilant of the source of the acoustic stimulus, which would alter their time budget. That is, during the time they are vigilant, they are not engaged in other behavior (Box BR1.4 of Figure 2); and (4) they may continue their pre-disturbance behavior and cope with the physiological consequences of continued exposure.

Marine animals might experience one of these behavioral responses, they might experience a sequence of several of these behaviors (for example, an animal might continue its pre-disturbance behavior for a period of time, then abandon an area after it experiences the consequences of physiological stress) or one of these behaviors might accompany responses such as permanent or temporary loss in hearing sensitivity. The narratives that follow summarize the information available on these behavioral responses.

BEHAVIORAL AVOIDANCE OF INITIAL EXPOSURES OR CONTINUED EXPOSURE (HORIZONTAL AND VERTICAL AVOIDANCE. As used in this Opinion, *behavioral avoidance* refers to animals that adjusts their spatial position relative to the source of the stimulus (in this case, sounds produced by a seismic survey vessel or the survey vessel itself) to avoid an adverse event, experience, or outcome they associate with the stimulus. Specifically, behavioral avoidance might consist of animals leaving an area ensonified by sounds produced by seismic surveys, avoid being exposed to the entire sound field produced by seismic surveys, or avoid being exposed to particular received levels within a sound field. We distinguish this from *evasion*, which occurs when an animal is already experiencing the adverse event, experience, or outcome. An evasive movement occurs when the animal adjusts its spatial position relative to the source of the stimulus to avoid continued exposure.

Richardson *et al*. (1995) noted that avoidance reactions are the most obvious manifestations of disturbance in marine mammals. There are few empirical studies of avoidance responses of free-living cetaceans to mid-frequency sonar. However, Kvadsheim *et al*. (2007) conducted a controlled exposure experiment in which killer whales (*Orcinus orca*) that had been fitted with D-tags were exposed to mid-frequency active sonar (Source A: was a 1.0 s upsweep 209 dB @ 1 - 2 kHz every 10 seconds for 10 minutes; Source B: was a 1.0 s upsweep 197 dB @ 6 - 7 kHz every 10 s for 10 min).

When exposed to Source A, a tagged killer whale and the group it was traveling with did not appear to avoid the source. When exposed to Source B, the tagged whales along with other whales that had been carousel feeding, ceased feeding during the approach of the sonar and moved rapidly away from the source (the received level associated with this response was not reported). When exposed to Source B, Kvadsheim and his co-workers reported that a tagged killer whale seemed to try to avoid further exposure to the sound field by immediately swimming away (horizontally) from the source of the sound; by engaging in a series of erratic and frequently deep dives that seemed to take it below the sound field; or by swimming away while engaged in a series of erratic and frequently deep dives. Although the sample sizes in this study are too small to support statistical analysis, the behavioral responses of the orcas were consistent with the results of other studies.

Maybaum (1993) conducted sound playback experiments to assess the effects of mid-frequency active sonar on humpback whales in Hawai'ian waters. Specifically, he exposed focal pods to sounds of a 3.3-kHz sonar pulse, a sonar frequency sweep from 3.1 to 3.6 kHz, and a control (blank) tape while monitoring the behavior, movement, and underwater vocalizations. The two types of sonar signals differed in their effects on the humpback whales, the whales exhibited avoidance behavior when exposed to both sounds. The whales responded to the pulse by increasing their distance from the sound source and responded to the frequency sweep by increasing their swimming speeds and track linearity.

In the Caribbean, sperm whales avoided exposure to mid-frequency submarine sonar pulses, ranging from 1,000 Hz to 10,000 Hz (IWC 2005). Sperm whales in the Gulf of Mexico seemed to avoid exposure to seismic surveys using mid- and low-frequency sound sources by increasing their distance from surveys vessles (Barkaszi et al 2012). Blue and fin whales have occasionally been reported in areas ensonified by airgun pulses; however, there have been no systematic analyses of their behavioral reactions to airguns. Sightings by observers on seismic vessels off the United Kingdom suggest that, at times of good sightability, the number of blue, fin, sei, and humpback whales seen when airguns are shooting are similar to the numbers seen when the airguns are not shooting (Stone 1997, 1998, 2000, 2001). However, fin and sei whale sighting rates were higher when airguns were shooting, which may result from their tendency to remain at or near the surface at times of airgun operation (Stone 2003). The analysis of the combined data from all years indicated that baleen whales stayed farther from airguns during periods of shooting (Stone 2003). Baleen whales also altered course more often during periods of shooting and more were headed away from the vessel at these times, indicating some level of localized avoidance of seismic activity (Stone 2003).

Sperm whales responded to military sonar, apparently from a submarine, by dispersing from social aggregations, moving away from the sound source, remaining relatively silent and becoming difficult to approach (Watkins *et al.* 1985). Brownell (2004) reported the behavioral responses of western gray whales off the northeast coast of Sakhalin Island to sounds produced by seismic activities in that region.

In 1997, the gray whales responded to seismic activities by changing their swimming speed and orientation, respiration rates, and distribution in waters around the seismic surveys. In 2001, seismic activities were conducted in a known feeding area of these whales and the whales left the feeding area and moved to areas farther south in the Sea of Okhotsk. They only returned to the feeding area several days after the seismic activities stopped. The potential fitness consequences of displacing these whales, especially mother-calf pairs and "skinny whales," outside of their the normal feeding area is not known; however, gray whales, like other large whales, must gain enough energy during the summer foraging season to last them the entire year. Sounds or other stimuli that cause them to abandon a foraging area for several days seems almost certain to disrupt their energetics and force them to make trade-offs like delaying their migration south, delaying reproduction, reducing growth, or migrating with reduced energy reserves.

Captive bottlenose dolphins and a white whale exhibited changes in behavior when exposed to 1 second pulsed sounds at frequencies similar to those emitted by the multi-beam sonar that is used by geophysical surveys (Ridgway *et al.* 1997, Schlundt *et al.* 2000), and to shorter broadband pulsed signals (Finneran *et al.* 2000, 2002). Behavioral changes typically involved what appeared to be deliberate attempts to avoid a sound exposure or to avoid the location of the exposure site during subsequent tests (Schlundt *et al.* 2000, Finneran *et al.* 2002). Dolphins exposed to 1-sec intense tones exhibited short-term changes in behavior above received sound levels of 178 to 193 dB re 1 μPa rms and belugas did so at received levels of 180 to 196 dB and above. Received levels necessary to elicit such responses to shorter pulses were higher (Finneran *et al.* 2000, 2002). Test animals sometimes vocalized after exposure to pulsed, mid-frequency sound from a watergun (Finneran *et al.* 2002). In some instances, animals exhibited aggressive behavior toward the test apparatus (Ridgway *et al.* 1997, Schlundt *et al.* 2000). It is not clear whether or to what degree the responses of captive animals might be representative of the responses of marine animals in the wild. For example, wild cetaceans sometimes avoid sound sources well before they are exposed to received levels such as those used in these experiments. Further, the responses of marine animals in the wild may be more subtle than those described by Ridgway *et al.* (1997) and Schlundt *et al.* (2000).

Richardson *et al*. (1995a) and Richardson (1997, 1998) used controlled playback experiments to study the response of bowhead whales in Arctic Alaska. In their studies, bowhead whales tended to avoid drill ship noise at estimated received levels of 110 to 115 dB and seismic sources at estimated received levels of 110 to 132 dB. Richardson *et al*. (1995) concluded that some marine mammals would tolerate continuous sound at received levels above 120 dB re 1 μPa for a few hours. These authors concluded that most marine mammals would avoid exposures to received levels of continuous underwater noise greater than 140 dB when source frequencies were in the animal's most sensitive hearing range.

Several authors noted that migrating whales are likely to avoid stationary sound sources by deflecting their course slightly as they approached a source (LGL and Greenridge 1987 in Richardson *et al*. 1995). Malme *et al*. (1983, 1984) studied the behavioral responses of gray whales (*Eschrichtius robustus*) that were migrating along the California coast to various sound sources located in their migration corridor. The whales they studied showed statistically significant responses to four different underwater playbacks of continuous sound at received levels of approximately 120 dB. The sources of the playbacks were typical of a drillship, semisubmersible, drilling platform, and production platform.

Morton *et al*. (2004) exposed killer whales (*Orcinus orca*) to sounds produced by acoustic harassment devices (devices that were designed to harass harbor seals, source levels were 194 dB at 10 kHz re 1μPa at 1 meter). They concluded that observations of killer whales declined dramatically in the experimental area (Broughton Archipelago) during the time interval the harassment devices had been used (but not before or after the use). Other investigators have concluded that gray whales and humpback whales abandoned some of their coastal habitat in California and Hawai'i, respectively, because of underwater noise associated with extensive vessel traffic (Gard 1974, Reeves 1977, Salden 1988).

Nowacek *et al*. (2004) conducted controlled exposure experiments on North Atlantic right whales using ship noise, social sounds of con-specifics, and an alerting stimulus (frequency modulated tonal signals between 500 Hz and 4.5 kHz). Animals were tagged with acoustic sensors (D-tags) that simultaneously measured movement in three dimensions. Whales reacted strongly to alert signals at received levels of 133-148 dB SPL, mildly to conspecific signals, and not at all to ship sounds or actual vessels. The alert stimulus caused whales to immediately cease foraging behavior and swim rapidly to the surface.

Several studies have demonstrated that cetaceans will avoid human activities such as vessel traffic, introduced sounds in the marine environment, or both. Lusseau (2003) reported that bottlenose dolphins in Doubtful Sound, New Zealand, avoided approaching tour boats by increasing their mean diving interval. Male dolphins began to avoid tour boats before the boats were in visible range, while female dolphins only began to avoid the boats when the boats became intrusive (he attributed the differential responses to differences in energetics: the larger body size of male dolphins would allow them to compensate for the energy costs of the avoidance behavior more than female dolphins). Bejder *et al*. (2006) studied the effects of vessel traffic on bottlenose dolphins in Shark Bay, Australia, over three consecutive 4.5-year periods. They reported that the dolphins avoided the bay when two tour operators began to operate in the bay.

Marine mammals may avoid or abandon an area temporarily during periods of high traffic or noise, returning when the source of the disturbance declines below some threshold (Lusseau 2004, Allen and Read 2000). Alternatively, they might abandon an area for as long as the disturbance persists. For example, Bryant *et al*. (1984 *in* Polefka 2004) reported that gray whales abandoned a calving lagoon in Baja California, Mexico following the initiation of dredging and increase in small vessel traffic. After the noise-producing activities stopped, the cow-calf pairs returned to the lagoon; the investigators did not report the consequences of that avoidance on the gray whales. Gard (1974) and Reeves (1977) reported that underwater noise associated with vessel traffic had caused gray whales to abandon some of their habitat in California for several years. Salden (1988) suggested that humpback whales avoid some nearshore waters in Hawai'i for the same reason.

As Bejder *et al*. (2006 and 2009) argued, animals that are faced with human disturbance must evaluate the costs and benefits of relocating to alternative locations; those decisions would be influenced by the availability of alternative locations, the distance to the alternative locations, the quality of the resources at the alternative locations, the conditions of the animals faced with the decision, and their ability to cope with or "escape" the disturbance (citing Beale and Monaghan 2004a, 2004b; Gill *et al*. 2001, Frid and Dill 2002, Lima and Dill 1990). Specifically, animals delay their decision to flee from predators and predatory stimuli that they detect, or until they decide that the benefits of fleeing a location are greater than the costs of remaining at the location or, conversely, until the costs of remaining at a location are greater than the benefits of fleeing (Ydenberg and Dill 1996). Ydenberg and Dill (1996) and Blumstein (2003) presented an economic model that recognized that animals will almost always choose to flee a site over some short distance to a predator; at a greater distance, animals will make an economic decision that weighs the costs and benefits of fleeing or remaining; and at an even greater distance, animals will almost always choose not to flee.

Based on a review of observations of the behavioral responses of 122 minke whales, 2,259 fin whales, 833 right whales, and 603 humpback whales to various sources of human disturbance, Watkins (1986) reported that fin, humpback, minke, and North Atlantic right whales ignored sounds that occurred at relatively low received levels, that had the most energy at frequencies below or above their hearing capacities appeared not to be noticed, or that were from distant human activities, even when those sounds had considerable energies at frequencies well within the whale's range of hearing. Most of the negative reactions that had been observed occurred within 100 m of a sound source or when sudden increases in received sound levels were judged to be in excess of 12 dB, relative to previous ambient sounds

From these observations, we would have to conclude that the distance between marine mammals and a source of sound, as well as the received level of the sound itself, will help determine whether individual animals are likely to respond to the sound and engage in avoidance behavior. At the limits of the range of audibility, endangered and threatened marine mammals are likely to ignore cues that they might otherwise detect. At some distance that is closer to the source, endangered marine mammals may be able to detect a sound produced by seismic and HRG survey activities, but they would not devote attentional resources to the sound (that is, they would filter it out as background noise or ignore it). We would not expect endangered marine mammals that find themselves within a sound field produced by a seismic shot or a HRG sonar ping where received levels might be as high as 140 dB to devote attentional resources to that stimulus because those individuals are more likely to be focusing their attention on stimuli and environmental cues that are considerably closer, even if they were aware of the signal.

Those animals that are closer to the source and not engaged in activities that would compete for their attentional resources (for example, mating or foraging) might engage in low-level avoidance behavior (changing the direction or their movement to take them away from or tangential to the source of the disturbance) possibly accompanied by short-term vigilance behavior, but they are not likely to change their behavioral state (that is, animals that are foraging or migrating would continue to do so). For example, we would expect endangered or threatened marine mammals that find themselves within a sound field produced by a seismic or HRG survey activities where received levels might range from 140 and 150 dB to engage in low-level avoidance behavior or short-term vigilance behavior, but they are not likely to change their behavioral state as a result of that exposure. The 150 dB isopleths for seismic and HRG equipment ranged from further than 20 km from the 5,400 in^3 airgun array, 6.4 km from the 90 in^3 array, 3.4 km from the chirp sub-bottom profiler or 8.2 km from the boomer or sparker.

At some distance that is closer still, these species are likely to engage in more active avoidance behavior followed by subsequent low-level avoidance behavior that does not bring them closer to the training activity. At the closest distances, we assume that endangered and threatened marine mammals would engage in vertical and horizontal avoidance behavior unless they have a compelling reason to remain in a location (for example, to feed). In some circumstances, this would involve abrupt vertical or horizontal movement accompanied by physiological stress responses. Within and adjacent to the Mid-Atlantic and South-Atlantic Planning Areas, we would expect these kind of responses at distances between 0 and 21 km for the 5,400 in^3 airgun array, 0.186 km for the 90 in^3 airgun array for seismic and HRG surveys for the oil and gas program and for HRG surveys for the renewable energy and marine minerals programs 0.045 km (for the boomer and sparker) and 0.042 km (for the chirp subbottom profiler), where received levels from these acoustic sources would equal or exceed 180 dB. However, at these distances endangered or threatened marine mammals would be aware of a wide array of visual and acoustic cues associated with seismic survey vessels (including sound associated with a ship's engines, the bow wake, etc.) and an animal's decision to change its behavior might be a response to an airgun shot, one of these other cues, or the entire suite of cues.

The evidence available also suggests that marine mammals might experience more severe consequences if an acoustic cue associated with a seismic or HRG survey leads them to perceive they face an imminent threat, but circumstances do not allow them to avoid or "escape" further exposure. At least six circumstances might prevent an animal from escaping further exposure to a seismic survey and could produce any of one the following outcomes:

1. when swimming away (an attempted "escape") brings marine mammals into a shallow coastal feature that causes them to strand;
2. they cannot swim away because the exposure occurred in a coastal feature that leaves marine mammals no "escape" route (for example, a coastal embayment or fjord that surrounds them with land on three sides, with the sound field preventing an "escape");
3. they cannot swim away because the marine mammals are exposed to multiple sound fields in a coastal or oceanographic feature that act in concert to prevent their escape;
4. they cannot dive "below" the sound field while swimming away because of shallow depths;
5. to remain "below" the sound field, they must engage in a series of very deep dives with interrupted attempts to swim to the surface (which might lead to pathologies similar to those of decompression sickness);
6. any combination of these phenomena.

Because seismic surveys are generally conducted in deeper waters well away from the coastline, endangered or threatened species are not likely to be exposed to sounds produced by seismic surveys under any of these conditions. For the HRG surveys in which acoustic sources can be heard by endangered or threatened species, the more limited sound fields and greater attenuation in more coastal environments should allow these animals to avoid or "escape" further exposure.

VIGILANCE. Attention is the cognitive process of selectively concentrating on one aspect of an animal's environment while ignoring other things (Posner 1994). Because animals (including humans) have limited cognitive resources, there is a limit to how much sensory information they can process at any time. The phenomenon called "attentional capture" occurs when a stimulus (usually a stimulus that an animal is not concentrating on or attending to) "captures" an animal's attention. This shift in attention can occur consciously or unconsciously (for example, when an animal hears sounds that it associates with the approach of a predator) and the shift in attention can be sudden (Dukas 2002, van Rij 2007). Once a stimulus has captured an animal's attention, the animal can respond by ignoring the stimulus, assuming a "watch and wait" posture, or treat the stimulus as a disturbance and respond accordingly, which includes scanning for the source of the stimulus or "vigilance" (Cowlishaw *et al*. 2004).

Vigilance is normally an adaptive behavior that helps animals determine the presence or absence of predators, assess their distance from conspecifics, or to attend to cues from prey (Bednekoff and Lima 1998, Treves 2000). Despite those benefits, however, vigilance has a cost of time: when animals focus their attention on specific environmental cues, it is not attending to other activities such as foraging. These costs have been documented best in foraging animals, where vigilance has been shown to substantially reduce feeding rates (Saino 1994, Beauchamp and Livoreil 1997, Fritz *et al*. 2002).

Animals will spend more time being vigilant, which translates into less time foraging or resting, when disturbance stimuli approach them more directly, remain at closer distances, have a greater group size (for example, multiple surface vessels), or when they co-occur with times that an animal perceives increased risk (for example, when they are giving birth or accompanied by a calf). Most of the published literature, however, suggests that direct approaches will increase the amount of time animals will dedicate to being vigilant. For example, bighorn sheep and Dall's sheep dedicated more time being vigilant, and less time resting or foraging, when aircraft made direct approaches over them (Frid 2001, Stockwell *et al*. 1991*)*.

Several authors have established that long-term and intense disturbance stimuli can cause population declines by reducing the body condition of individuals that have been disturbed, followed by reduced reproductive success, reduced survival, or both (Daan *et al*. 1996, Madsen 1994, White 1983). For example, Madsen (1994) reported that pink-footed geese (*Anser brachyrhynchus*) in undisturbed habitat gained body mass and had about a 46% reproductive success compared with geese in disturbed habitat (being consistently scared off the fields on which they were foraging) which did not gain mass and has a 17% reproductive success. Similar reductions in reproductive success have been reported for mule deer (*Odocoileus hemionus*) disturbed by all-terrain vehicles (Yarmoloy *et al*. 1988), caribou disturbed by seismic exploration blasts (Bradshaw *et al*. 1998), caribou disturbed by low-elevation military jet-fights (Luick *et al*. 1996), and caribou disturbed by low-elevation jet flights (Harrington and Veitch 1992). Similarly, a study of elk (*Cervus elaphus*) that were disturbed experimentally by pedestrians concluded that the ratio of young to mothers was inversely related to disturbance rate (Phillips and Alldredge 2000).

The primary mechanism by which increased vigilance and disturbance appear to affect the fitness of individual animals is by disrupting an animal's time budget and, as a result, reducing the time they might spend foraging and resting (which increases an animal's activity rate and energy demand). For example, a study of grizzly bears (*Ursus horribilis*) reported that bears disturbed by hikers reduced their energy intake by an average of 12 kcal/min (50.2 x 103kJ/min), and spent energy fleeing or acting aggressively toward hikers (White *et al*. 1999).

CONTINUED PRE-DISTURBANCE BEHAVIOR, HABITUATION, OR NO RESPONSE. Under some circumstances, some individual animals that would be exposed to seismic and electromechanical transmissions (for sources within the range of audibility of listed species in this Opinion) and other sounds associated with seismic and HRG survey activities will continue the behavioral activities they were engaged in before they were exposed (Richardson *et al*. 1995). For example, Watkins (1986) reviewed data on the behavioral reactions of fin, humpback, right and minke whales that were exposed to continuous, broadband low-frequency shipping and industrial noise in Cape Cod Bay. He concluded that underwater sound was the primary cause of behavioral reactions in these species of whales and that the whales responded behaviorally to acoustic stimuli within their respective hearing ranges. Watkins also noted that whales showed the strongest behavioral reactions to sounds in the 15 Hz to 28 kHz range, although negative reactions (avoidance, interruptions in vocalizations, etc.) were generally associated with sounds that were either unexpected, too loud, suddenly louder or different, or perceived as being associated with a potential threat (such as an approaching ship on a collision course). In particular, whales seemed to react negatively when they were within 100 m of the source or when received levels increased suddenly in excess of 12 dB relative to ambient sounds. At other times, the whales ignored the source of the signal and all four species habituated to these sounds.

Nevertheless, Watkins concluded that whales ignored most sounds in the background of ambient noise, including the sounds from distant human activities even though these sounds may have had considerable energies at frequencies well within the whale's range of hearing. Further, he noted that fin whales were initially the most sensitive of the four species of whales, followed by humpback whales; right whales were the least likely to be disturbed and generally did not react to low-amplitude engine noise. By the end of his period of study, Watkins (1986) concluded that fin and humpback whales had generally habituated to the continuous, broad-band, noise of Cape Cod Bay while right whales did not appear to change their response.

Aicken *et al*. (2005) monitored the behavioral responses of marine mammals to a new low-frequency active sonar system that was being developed for use by the British Navy. During those trials, fin whales, sperm whales, Sowerby's beaked whales, long-finned pilot whales (*Globicephala melas*), Atlantic white-sided dolphins, and common bottlenose dolphins were observed and their vocalizations were recorded.

These monitoring studies detected no evidence of behavioral responses that the investigators could attribute to exposure to the low-frequency active sonar during these trials (some of the responses the investigators observed may have been to the vessels used for the monitoring).

There are several reasons why such animals might continue their pre-exposure activity:

1. RISK ALLOCATION. When animals are faced with a predator or predatory stimulus, they consider the risks of predation, the costs of anti-predator behavior, and the benefits of continuing a pre-existing behavioral pattern when deciding which behavioral response is appropriate in a given circumstance (Bejder *et al*. 2008, Gill *et al*. 2001, (Houston and McNamara 1986, Lima 1998, Lima and Bednekoff 1999, Ydenberg and Dill 1996). Further, animals appear to detect and adjust their responses to temporal variation in predation risks (Kat and Dill 1998, Lima and Bednekoff 1999, Rodriguez-Prieto *et al*. 2008). As a result, animals that decide that the ecological costs of changing their behavior exceeds the benefits of continuing their behavior, would be expected to continue their pre-existing behavior. For example, baleen whales, which only feed during part of the year and must satisfy their annual energetic needs during the foraging season, are more likely to continue foraging in the face of disturbance. Similarly, a cow accompanied by her calf is less likely to flee or abandon an area at the cost of her calf's survival.

   This does not mean, however, that there are no costs involved with continuing pre-disturbance behavior in the face of predation or disturbance. When animals make risk allocation decisions, they tolerate some exposure to a stressor, which means they accept some risk. We assume that individual animals that are exposed to sounds associated with seismic and HRG survey activities will apply the economic model we discussed earlier (Ydenberg and Dill 1996). By extension, we assume that animals that choose to continue their pre-disturbance behavior would have to cope with the costs of doing so, which will usually involve physiological stress responses and the energetic costs of stress physiology (Frid and Dill 2002).

2. HABITUATION. When free-ranging animals do not appear to respond when presented with a stimulus, they are commonly said to have become habituated to the stimulus (Bejder *et al*. 2008, Rodriguez-Prieto *et al*. 2008, and the example cited earlier from Watkins 1986). Habituation has been given several definitions, but we apply the definition developed by Thompson and Spencer (1966) and Groves and Thompson (1970), which are considered classic treatments of the subject, as modified by Rankin *et al*. (2009): *an incremental reduction in an animal's behavioral response to a stimulus that results from repeated stimulation to that stimulus and that does not*

involve sensory adaptation, sensory fatigue, or motor fatigue. The value of this definition, when compared with other definitions (for example, Bejder *et al*. 2009 citing Thorpe 1963), is that it would lead us to establish that an animal did not experience reduced sensory sensitivity to a stimulus (which would be accompanied by threshold shifts, for example) before we would conclude that the animal had become habituated to the stimulus. Habituation has been traditionally distinguished from sensory adaptation or motor fatigue using dishabituation (presentation of a different stimulus that results in an increase of the decremented response to the original stimulus), by demonstrating stimulus specificity (the response still occurs to other stimuli), or by demonstrating frequency dependent spontaneous recovery (more rapid recovery following stimulation delivered at a high-frequency than following stimulation delivered at a low frequency).

Animals are more likely to habituate (and habituate more rapidly) to a stimulus, the less intense the stimulus (Rankin *et al*. 2009). Conversely, numerous studies suggest that animals are less likely to habituate (that is, exhibit no significant decline in their responses) as the intensity of the stimulus increases (Rankin *et al*. 2009). Further, after animals have become habituated to a stimulus, their responses to that stimulus recover (a process that is called “spontaneous recovery”) over time, although habituation becomes more rapid and pronounced after a series of habituation-recovery events (a process that is called “potentiation of habituation”).

3 The individuals that might be exposed may have lowered sensitivity to the stimulus. This might occur because the animals are naïve to the potential risks associated with seismic and HRG survey activities (which would be more common among juveniles than adults) or they have limited sensory sensitivity by physiological constitution or constitutional endowment.

The results reported by Watkins (1986) and Aicken *et al*. (2005) could be explained either by concluding that the marine mammals had habituated to the sounds or they could be explained by concluding that the animals had made a decision to continue their pre-disturbance behavior despite the potential risks represented by the sounds (that is, the animals tolerated the disturbance). The results reported by Watkins (1986) are better explained using risk allocation than habituation because he associated the strongest, negative reactions (avoidance, interruptions in vocalizations, etc.) with sounds that were either unexpected, too loud, suddenly louder or different, were perceived as being associated with a potential threat (such as an approaching ship on a collision course), or were from distant human activities despite having considerable energy at frequencies well within the whale's range of hearing (whales would be less likely to respond to cues they would associate with a predator if their distance from the predator preserved their ability to escape a potential attack).

Responses of North Atlantic Right Whale cow/calf pairs to HRG Surveys within Critical Habitat

Critical habitat off the southeastern U.S. coast has the lowest ambient sound level amongst three areas investigated within the range of North Atlantic right whales (72-85 dB re: 1 µPa/Hz over the frequency range of 50-350 Hz, with peak frequencies on the bottom of the spectrum and variation of 30 dB within 24 hours) (Parks et al. 2009). Right whales may adjust their vocal behavior based upon this ambient sound field (Parks et al. 2009). This suggests the possibility of masking issues for North Atlantic right whales, even in relatively quiet areas.

Data on the impacts of the proposed activities on right whales, particularly acoustic sources (excluding vessel noise) that may reduce the quality of critical habitat are few (Doug Nowacek, Duke University, pers. comm., 2013). Southern right whales moved away from 200 Hz tonal signals, humpback whale playbacks, as well as water noise when played from a loudspeaker (Clark and Clark 1980). However, ensonification of right whales within 100 m using a sonar transmitting at frequencies greater than 80 kHz did not elicit responses (Miller and Potter 2000). A seminal manuscript by Nowacek et al. (2004) found that North Atlantic right whales, carrying tags that record received sound levels and whale movements, responded strongly to a signal designed to be unique and audible to right whales at received levels of 134-148 dB re 1 $\mu Pa{\cdot}m_{rms}$, but did not respond to vessel noise. The alert signal contained three different components: alternating 1 s pure tones at 500 and 850 Hz; a 2 s logarithmic down-sweep from 4,500 to 500 Hz; and a pair of low-high (1,500 and 2,000 Hz) sine wave tones amplitude modulated at 120 Hz and each 1 s long. It is unknown which of these signals (individually or in combination) whales responded to. Responses included cessation of foraging, low-angle, high speed ascent, and remaining at or near the surface for the duration of the alert signal exposure (18 min). A North Pacific right whale was observed from 1 km away, approaching to within 400 m while a 40 in^3 airgun (modeled to have a 160 dB re 1 $\mu Pa{\cdot}m_{rms}$ radius of 385 m) was discharging before being recognized as a right whale and the airgun was shutdown (Davis et al. 2011). The individual did not appear to respond to the vessel or its acoustic sources, but continued feeding behavior from the time of first sighting to when it could no longer be observed (Davis et al. 2011). A sub-bottom profiler (3.5 kHz dominant frequency, maximum source level of 204 dB re 1 µPa·m) and multibeam echosounder (10.5-13 kHz, maximum source level of 242 dB re 1 $\mu Pa{\cdot}m_{rms}$) were operating at the time of the encounter (Meagan Cummings, Lamont-Doherty Earth Observatory, pers. comm., 2013). However, environmental noise at the time was likely high (Beaufort 4 conditions)(Davis et al. 2011). Dr. Peter Best (University of Pretoria, pers. comm., 2013) suggested high levels of seismic survey activity associated with oil and gas exploration may be leading to a shift in non-mother-calf groups from South African waters to Namibia, although he could not discount this as a natural re-distribution to a former wintering site.

Responses of right whales to vessel approaches (and the probable acoustic stressors they present) have been described in more detail, particularly in southern right whales nursery habitat. Slow, standardized approaches of small vessels to southern right whales in their nursery habitat off Argentina while in resting, traveling, or socializing behavioral states and solitary, surface active, and non-surface active groups produced mixed results (Alejandro and Els 2010). Time spent socializing decreased notably as did aerial behavior (from 21 to 2%) (Alejandro and Els 2010). Traveling appeared undisturbed, but traveling individuals doubled their time spent resting after approaches stopped (Alejandro and Els 2010). Mother-calf pairs were specifically avoided in the study (Alejandro and Els 2010). Another study in the same area found very close boat approaches and swimmers entering the water with southern right whales resulted in decreases in resting, socializing, and surface active behavior, while traveling and reorientation increased (Lundquist et al. 2012). In 17% of cases (27% for mother-calf pairs), boats could not approach closely enough to put swimmers in the water due to whale evasion, with mother-calf pairs typically swimming away quickly, reorienting, and staying underwater for relatively long periods (Lundquist et al. 2012). Mother-calf pairs showed the strongest responses of all group and sex categories, avoiding the boat and swimmers 61% of the time (Lundquist et al. 2012). Faster swimming was first observed when the vessel approached to within 500 m (Lundquist et al. 2012). Individuals and groups frequently returned to baseline behavior after the vessel's departure (Lundquist et al. 2012). Rivarola et al. (2001) hypothesized that the impacts of extensive, impactful whale watching on southern right whales in Argentina may have contributed to mothers switching to a different habitat 30-40 km away, although movement between years and disparate nursery grounds has been documented (Rowntree et al. 2001), and whale numbers appearing in the whale watching areas corresponded with increasing whale watch activity (Rivarola et al. 2001). Vermuelen et al. (2012) found that southern right whales in Argentina shortened social interactions 13% when vessels were near (following slow, controlled approaches and frequent swimmer deployment), while propensity to travel increased and whales rested less. These responses ended shortly after close vessel approaches ended (Vermeulen et al. 2012). Approaches of mother-calf pairs on a Brazilian southern right whale nursery in which whale watching only recently began found responses to be generally "uninterested", although more than a third of individuals approached the vessel and 10% displayed a "negative" response (Groch 2001). Southern right whales in South African nursery habitat approached vessels approaching them in almost half of cases, with idling motors possibly attracting individuals (Findlay 1999). Swimming speeds appeared to change from baseline only when active pursuit of right whales occurred (Findlay 1999).

North Atlantic right whales may not respond at all to kayaks, sailing sloops, or steel-hulled diesel-powered vessels approaching within five meters, although other individuals (possibly under different contexts) have responded to the same diesel-powered vessel from 50 m away, usually by turning away from the path of the ship (Goodyear 1993). Baumgartner and Mate (2003) found that 71% of 42 North Atlantic right whales approached (and sometimes tagged) in a rigged inflatable boat within 10 m did not overtly respond. Of those that did respond, behaviors included head lifts and lunges, back arching, rolling, and fluke beats. Feeding dive durations were also shorter by 13-17% in the dive following approach/tagging, but no difference was found in the duration of subsequent dives. Mate et al. (1997) found that although North Atlantic right whales generally responded to and avoided close approach, the level of response varied. Watkins (1986) found that whales are more responsive to approach when they are inactive and less responsive when feeding or socializing. Researcher effort in North Atlantic right whale critical habitat supports a variety of responses to closely approaching small vessels, ranging from immediate evasion to non-response, with most responses in between consisting of initial tolerance followed by eventual slow movement away (Clay George, George Department of Natural Resources, pers. comm. 2013). Cumulative annual report data for a variety of North Atlantic right whale researchers (including Mr. George) utilizing diverse methodologies between 1995 and 2010 and over 5,000 vessel approaches support an overall response rate of roughly 4%. Reductions in ambient sound associated with vessel noise were associated with reductions in stress hormone metabolites in North Atlantic right whales (Rolland et al. 2012).

Non-acoustic stressors can help us understand right whale responses to anthropogenic stressors in general. Responses of southern right whales to biopsy were equal for males and females, but singletons were more responsive than groups (Best et al. 2005). However, cows with calves were more responsive than singletons. Cows appeared to sensitive to repeated biopsy over the course of two months, but calves were not (Best et al. 2005). Responses consisted of minor, ephemeral behavioral changes (Best et al. 2005). Similar responses were found in North Atlantic right whales, which responded 19% of biopsy attempts (Brown et al. 1991). Reeb and Best (2006) also documented generally no or low- to moderate-level responses of right whales to pole biopsy techniques. Follow-up work over the following decade failed to identify any change in calf survival or intercalving interval for cows, although power to detect changes was low (Best et al. 2005).

Sea Turtles

Available studies suggest some sea turtles exhibit an avoidance reaction to airgun-generated sounds. McCauley *et al.* (2000b; 2000a) investigated the effects of airguns on sea turtle behavior. The authors found that green and loggerhead sea turtles show avoidance to airgun arrays at 2 km (1.1 nm) and at 1 km (0.54 nm) with received levels of 166 dB re 1 µPa and 175 dB re 1 µPa, respectively. Individual sea turtles responded consistently by noticeably increasing swimming activity above a level of approximately 166 dB re 1 µPa (rms), as compared to swimming during non-airgun operation periods. The increase in swimming behavior tracked the received airgun level, by increasing at increasing levels. Above 175 dB re 1 µPa, turtle behavior became more erratic, possibly indicating the turtles were in an agitated state. In studies by Lenhardt (1994) and Lenhardt *et al.* (1983), loggerhead and Kemp's ridley turtles responded to airgun pulses and low-frequency sound – e.g., by becoming active and swimming to the surface upon exposure. Moein *et al*. (1994) used an evoked potential method to test sea turtle hearing and reported avoidance behavior in loggerhead sea turtles at the beginning of airgun exposure trials; however, repeated airgun exposures days after the initial tests did not elicit a statistically significant avoidance response. The authors concluded this may be due to either habituation or temporary threshold shift in the turtles hearing capability. Based on a review of sea turtle data from 11 L-DEO seismic surveys since 2003, Holst *et al*. (2006) concluded that turtles exhibited localized avoidance during both large- and small-source seismic surveys.

Although studies suggest sea turtles are most likely to avoid seismic airgun pulses, monitoring reports from seismic surveys indicate occasions when sea turtles were likely exposed to seismic airgun pulses – e.g., green, leatherback, and olive ridley turtles during surveys in the Hess Deep area of the eastern tropical Pacific Ocean (Smultea and Holst 2003 as cited in NMFS, 2006h). Of six sea turtles that were sighted during those surveys, five were seen while airguns were active. In at least one instance, an olive ridley sea turtle was sighted within ten meters of the array while active. This turtle was reported to exhibit visible responses to either exposure to the seismic pulses or to the physical presence of the array and floats. Although the turtle swam away from the vessel and was not reported to suffer physical injury, it is assumed the turtle experienced a stress response to its exposure that may have risen to the level of harassment. Similarly, Holst *et al.* (2005b) report during seismic operations in the southern Gulf of Mexico off the Yucatán Peninsula, that seven sea turtles were sighted within the 180 dB safety radius in shallow water (< 40 m). One of these turtles was reported to be actively swimming away from the seismic source. Six of these turtles were seen < 200 m from the operating airguns before the airguns were powered- or shut-down (Holst, et al., 2005b).

A more recent study by DeRuiter and Doukara (2012) reported on the observed reactions of loggerhead sea turtles during a seismic survey in the Mediterranean Sea off Algeria. The seismic survey was conducted with a 13-airgun array (nominal source level of 252 dB re 1µPa at 1 m (peak)), towed at a depth of 11.5m at 5 kts. All but 3 (swimming underwater) of the 164 loggerheads observed (98 percent), were motionless at the surface when observed and ranged from 10s of meters to nearly 2 km from the vessel when sighted. Observed reactions included head raising (6), flipper agitation (7), air bubble production around head (2) or diving. Of the 86 turtles whose dive behavior could be visually tracked until their passage more than 100 m behind the array, 49 (57 percent) dove (dove before passing behind the airgun array) and 37 (43 percent) did not dive (were in view at the surface until they had passed more than 100 m behind the airgun array). At least 6 turtles dove immediately following an airgun shot indicating a startle response. The authors suggest that loggerhead turtles interrupted basking behavior and dove (an avoidance response) in response to airgun sounds at relatively close ranges (130 m median distance, 839 m maximum distance) and exposure levels estimated at about 191 dB re 1µPa at 1 m (peak) at 130 m and 175 dB re 1µPa at 1 m (peak) at 839 m.

As noted above for marine mammals, Bejder *et al*. (2006 and 2009) argued, animals that are faced with human disturbance must evaluate the costs and benefits of relocating to alternative locations; those decisions would be influenced by the availability of alternative locations, the distance to the alternative locations, the quality of the resources at the alternative locations, the conditions of the animals faced with the decision, and their ability to cope with or “escape” the disturbance (citing Beale and Monaghan 2004a, 2004b; Gill *et al*. 2001, Frid and Dill 2002, Lima and Dill 1990). Animals delay their decision to flee from predators and predatory stimuli that they detect, or until they decide that the benefits of fleeing a location are greater than the costs of remaining at the location or, conversely, until the costs of remaining at a location are greater than the benefits of fleeing (Ydenberg and Dill 1996). Ydenberg and Dill (1996) and Blumstein (2003) presented an economic model that recognized that animals will almost always choose to flee a site over some short distance to a predator; at a greater distance, animals will make an economic decision that weighs the costs and benefits of fleeing or remaining; and at an even greater distance, animals will almost always choose not to flee.

Sea turtle hearing thresholds appear to be higher than those for mammals and DFO (2004) concluded it is unlikely that sea turtles would be more sensitive to seismic operations than cetaceans, based on available studies. Sea turtles are expected to be less sensitive to sounds; however, behavioral responses to environmental sounds are documented in several controlled experiments. Some possible reactions to low frequency sounds include startle responses and rapid swimming (M. Lenhardt, 2002; McCauley 2001 as cited in NMFS, 2006h), as well as swimming towards the surface at the onset of the sound (M. L. Lenhardt, 1994).

Because it would be difficult to distinguish between animals that continue their pre-disturbance behavior when exposed to seismic and HRG transmissions because of a risk-decision and animals that habituate to disturbance (that is, they may have experienced low-level stress responses initially, but those responses abated over time), we do not assume that endangered or threatened sea turtles that do not appear to respond to sounds produced by equipment employed during seismic and HRG surveys will become habituated to those sounds. Without more evidence of actual habituation, such an assumption would lead us to fail to protect these species when protection was warranted.

Atlantic Sturgeon

Sturgeon in general can be classified as fishes that detect sounds from below 50 Hz to perhaps 800-1,000 Hz (though several probably only detect sounds to 600-800 Hz). These fishes have a swim bladder but no known structures in the auditory system that would enhance hearing, and sensitivity (lowest sound detectable at any frequency) is not very great. Sounds would have to be more intense to be detected compared to fishes with swim bladders that enhance hearing. Sturgeon can detect both particle motion and pressure.

Impaired Communication

Communication is an important component of the daily activity of animals and ultimately contributes to their survival and reproductive success. Animals communicate to find food (Elowson *et al*. 1991, Marler *et al*. 1986, Stokes 1971), acquire mates (Patricelli *et al*. 2002, Ryan 1985, Stokes 1971), assess other members of their species (Owings *et al*. 2002, Parker 1974, Sullivan 1984), evade predators (Greig-Smith 1980, Marler 1955, Vieth *et al*. 1980), and defend resources (Alatalo *et al*. 1990, Falls 1963, Zuberbuehler *et al*. 1997). Human activities that impair an animal's ability to communicate effectively might have significant effects on the animals experiencing the impairment.

Communication usually involves individual animals that produce vocalizations or visual or chemical displays for other individuals. Masking, which we discuss separately (below), affects animals that are trying to receive acoustic cues in their environment, including vocalizations from other members of the animals' species or social group (Dunlop *et al*. 2010). However, anthropogenic noise presents separate challenges for animals that are vocalizing. This subsection addresses the probable responses of individual animals whose attempts to vocalize or communicate are affected by seismic and HRG transmissions.

When they vocalize, animals are aware of environmental conditions that affect the "active space" of their vocalizations, which is the maximum area within which their vocalizations can be detected before it drops to the level of ambient noise (Brenowitz 2004, Brumm *et al*. 2004, Lohr *et al*. 2003). Animals are also aware of environment conditions that affect whether listeners can discriminate and recognize their vocalizations from other sounds, which are more important than detecting a vocalization (Brenowitz 1982, Brumm *et al*. 2004, Dooling 2004, Dunlop *et al*. 2010, Marten and Marler 1977, Patricelli *et al*. 2006).

Most animals that vocalize have evolved with an ability to make vocal adjustments to their vocalizations to increase the signal-to-noise ratio, active space, and recognizability of their vocalizations in the face of temporary changes in background noise (Brumm *et al*. 2004, Cody and Brown 1969, Dunlop *et al*. 2010, Patricelli *et al*. 2006). Vocalizing animals will make one or more of the following adjustments to preserve the active space and recognizability of their vocalizations:

1. Adjust the amplitude of vocalizations (Box BR2.1 of Figure 3). Animals responding in this way increase the amplitude or pitch of their calls and songs by placing more energy into the entire vocalization or, more commonly, shifting the energy into specific portions of the call or song.

This response is called the "Lombard reflex" or "Lombard effect" and represents a short-term adaptation to vocalizations in which a signaler increases the amplitude of its vocalizations in response to an increase in the amplitude of background noise (Lombard 1911). This phenomenon has been studied extensively in humans, who raise the amplitude of the voices while talking or singing in the face of high, background levels of sound (Lombard 1911, Dunlop *et al*. 2010, Tonkinson 1990).

Other species experience the same phenomenon when they vocalize in the presence of high levels of background sound. Brumm (2004) studied the songs of territorial male nightingales (*Luscinia megarhynchos*) in the city of Berlin, Germany, to determine whether and to what degree background noise (from automobile traffic) produced a Lombard effect in these birds. Based on his studies, the birds increased the volume of their songs in response to traffic noise by 14 dB (their songs were more than 5 times louder than birds vocalizing in quiet sites). Cynx *et al*. (1998) reported similar results based on their study of zebra finches (*Taeniopygia guttata*) exposed to white noise.

Although this type of response also has not been studied extensively in marine animals, Holt *et al*. (2007) reported that endangered southern resident killer whales (*Orcinus orca*) in Haro Strait off the San Juan Islands in Puget Sound, Washington, increased the amplitude of their social calls in the face of increased sounds levels of background noise.

2. Adjust the frequency structure of vocalizations (Box BR2.2 of Figure 3). Animals responding in this way adjust the frequency structure of their calls and songs by increasing the minimum frequency of their vocalizations while maximum frequencies remain the same. This reduces the frequency range of their vocalizations and reduces the amount of overlap between their vocalizations and background noise.

Slabbekorn and Ripmeister (2008), Slabbekorn and den Boer-Visser (2006), and Slabbekorn and Peet (2003) studied patterns of song variation among individual great tits (*Parus major*) in an urban population in Leiden, The Netherlands, and among 20 different urban and forest populations across Europe and the United Kingdom. Adult males of this species that occupied territories with more background noise (primarily traffic noise) sang with higher minimum frequencies than males occupying non-urban or quieter sites. Peak or maximum frequencies of these songs did not shift in the face of high background noise.

3. Adjust temporal structure of vocalizations (Box BR2.3 of Figure 3). Animals responding this way adjust the temporal structure of their vocalizations by changing the timing of modulations, notes, and syllables within vocalizations or increasing the duration of their calls or songs.

Cody and Brown (1969) studied the songs of adult male Bewick wrens and wrentits that occupied overlapping territories and whose songs had similar physical characteristics (similar song lengths, frequency structure, and amplitude). They reported that wrentits adjusted the timing of their songs so they occurred when the songs of the Bewick wrens subsided.

Ficken *et al*. (1974) studied vocalizations of ten red-eyed vireos (*Vireo olivaceus*) and least flycatchers (*Empidonax minimus*) at Lake Itasca, Minnesota (a total of 2283 songs). They reported that flycatchers avoided acoustic interference from red-eyed vireos by inserting their shorter songs between the longer songs of the vireos. Although there is some mutual avoidance of acoustic interference, the flycatcher tends more strongly to insert its short songs in between the longer songs of the vireo rather than vice versa. Indeed, most of the overlap occurred when the flycatcher began singing just after the vireo had begun, suggesting that the flycatcher had not heard the vireo begin singing.

Two studies reported that some mysticete whales stopped vocalizing when exposed to active sonar. Miller *et al*. (2000) recorded the vocal behavior of singing humpback whales continuously for several hours using a towed, calibrated hydrophone array. They recorded at least two songs in which the whales were exposed to low-frequency active sonar transmissions (42 second signals at 6 minute intervals; sonar was broadcast so that none of the singing whales were exposed at received levels greater than 150 dB re

1µPa). They followed sixteen singing humpback whales during 18 playbacks. In nine follows, whales sang continuously throughout the playback; in four follows (or 0.2778; 95% CI: 0.1250 to 0.5087), the whale stopped singing when he joined other whales (a normal social interaction); and in five follows, the singer stopped singing, presumably in response to the playback. Of the six whales whose songs they analyzed in detail, songs were 29% longer, on average, during the playbacks. Song duration returned to normal after exposure, suggesting that the whale's response to the playback was temporary. Melcón *et al*. (2012) reported that during 110 of the 395 d-calls they recorded during mid-frequency active sonar transmissions, blue whales stopped vocalizing at received levels ranging from 85 to 145 dB, presumably in response to the sonar transmissions. The proportion of blue whales that stopped vocalizing during their study was 0.2785 (95%CI: 0.2366 to 0.3247). Combining the results of these two studies would lead us to expect 0.2784 (95%CI: 0.1800 to 0.4040) of mysticete vocalizations to stop when vocalizations coincide with active sonar transmissions. We assume the same proportion of whales would stop vocalizing when exposed to sounds produced by equipment employed during seismic and HRG surveys.

Foote *et al*. (2004) compared recordings of endangered southern resident killer whales that were made in the presence or absence of boat noise in Puget Sound during three time periods between 1977 and 2003. They concluded that the duration of primary calls in the presence of boats increased by about 15% during the last of the three time periods (2001 to 2003). They suggested that the amount of boat noise may have reached a threshold above which the killer whales need to increase the duration of their vocalization to avoid masking by the boat noise.

4. Adjust the temporal delivery of vocalizations (Boxes BR2.4 and BR2.5 of Figure 3). Animals responding in this way change when they vocalize or change the rate at which they repeat calls or songs.

For example, tawny owls (*Strix aluco*) reduce the rate at which they call during rainy conditions (Lengagne and Slater 2002). Brenowitz (1982) concluded that red-winged blackbirds (*Agelaius phoeniceus*) had the largest active space, or broadcast area, for their calls at dawn because of relatively low turbulence and background noise when compared with other times of the day. Brown and Handford (2003) concluded that swamp and white-throated sparrows (*Melospiza georgiana* and *Zonotrichia albicollis*, respectively) tended to sing at dawn, as opposed to other times of the day, because they encountered the fewest impediments to acoustic transmissions during that time of the day.

Many animals will combine several of these strategies to compensate for high levels of background noise. For example, Brumm *et al*. (2004) reported that common marmosets (*Callithrix jacchus*) increased the median amplitude of the twitter calls as well as the duration of the calls in response to increased background noise. King penguins (*Aptenodytes patagonicus*) increase the number of syllables in a call series and the rate at which they repeat their calls to compensate for high background noise from other penguins in a colony or high winds (Lengagne *et al*. 1999). California ground squirrels (*Spermophilus beecheyi*) shifted the frequencies of their alarm calls in the face of high ambient noise from highway traffic (Rabin *et al*. 2003). However, they only shifted the frequency of the second and third harmonic of these alarm calls, without changing the amount of energy in the first harmonic. By emphasizing the higher harmonics, the ground squirrels placed the peak energy of their alarm calls above the frequency range of the masking noise from the highway. Wood and Yezerinac (2006) reported that song sparrows (*Melospiza melodus*) increased the frequency of the lowest notes in their songs and reduced the amplitude of the low frequency range of their songs. Fernandez-Juricic *et al*. (2005) reported that house finches (*Carpodacus mexicanus*) adopted the same strategy to compensate for background noise.

Although this form of vocal adjustment has not been studied extensively in marine animals, Dahlheim (1987) studied the effects of man-made noise, including ship, outboard engine and oil-drilling sounds, on gray whale calling and surface behaviors in the San Ignacio Lagoon, Baja, California. She reported statistically significant increases in the calling rates of gray whales and changes in calling structure (as well as swimming direction and surface behaviors) after exposure to increased noise levels during playback experiments. Although whale responses varied with the type and presentation of the noise source, she reported that gray whales generally increased their calling rates, the level of calls received, the number of frequency-modulated calls, number of pulses produced per pulsed-call series and call repetition rate as noise levels increased. Melcón *et al*. (2012) reported that 0.4287 (95%CI: 0.3583 to 0.5020) of blue whale vocalizations (d-calls) stopped when the whales were exposed to ship noise off Southern California.

Parks *et al*. (2007) reported that surface active groups of North Atlantic right whales would adopt this strategy as the level of ambient noise increased. As ambient noise levels increased from low to high, the minimum frequency of right whale “scream calls” increased from 381.4 Hz (± 16.50), at low levels of ambient noise, to 390.3 Hz (± 15.14) at medium noise levels, to 422.4 Hz (± 15.55) at high noise levels. Surface active groups of North Atlantic right whales would also increase the duration and the inter-call interval of their vocalizations as the level of ambient noise increased. As noise levels increased from low to high, the duration of right whale “scream calls” would increase from 1.18 seconds (± 0.08) at low levels of ambient noise to 1.22 seconds (± 0.08) at high noise levels (durations decreased to 1.11 seconds ± 0.07 at medium noise levels). The inter-call intervals of these vocalizations would increase from 17.9 seconds (± 5.06) at low levels of ambient noise, to 18.5 seconds (± 4.55) at medium noise levels, to 28.1

seconds (± 4.63) at high noise levels. In a more recent study, Parks *et al* (2011) investigated the noise dependent amplitude modification of individual right whale calls. The study indicates that right whales were able to maintain communication in moderate noise levels (92-143 dB in 20 Hz-8 Hz frequency bands) measured from the tag.

FITNESS CONSEQUENCES OF VOCAL ADJUSTMENTS. Although the fitness consequences of these vocal adjustments remain unknown, like most other trade-offs animals must make, some of these strategies probably come at a cost (Dunlop *et al*. 2010, Patricelli *et al*. 2006). For example, vocalizing more loudly in noisy environments may have energetic costs that decrease the net benefits of vocal adjustment and alter the bird's energy budget (Brumm 2004, Wood and Yezerinac 2006). Lambrechts (1996) argued that shifting songs and calls to higher frequencies was also likely to incur energetic costs.

In addition, Patricelli *et al*. (2006) argued that females of many species use the songs and calls of males to determine whether a male is an appropriate potential mate (that is, the animal must recognize the singer as a member of their species); if males must adjust the frequency or temporal features of their vocalizations to avoid masking by noise, they may no longer be recognized by conspecific females (Brumm 2004, Slabbekoorn and Peet 2003, Wood and Yezerinac 2006). Although this line of reasoning was developed for bird species, the same line of reasoning should apply to marine mammals, particularly for species like fin and sei whales whose song structures appear to be very similar.

However, if an animal fails to make vocal adjustments in presence the of masking noise, that failure might cause the animal to experience reduced reproductive success or longevity because it fails to communicate effectively with other members of its species or social group, including potential mates.

Based on the evidence available, endangered blue, fin, humpback, north Atlantic right, sei and sperm whales may experience impaired communication because they vocalize at frequencies that overlap with those of the acoustic equipment employed during seismic surveys. As a result, we assume that some of the whales that are exposed to sound transmissions during seismic surveys might experience impaired communication as a result of that exposure. Sperm whales may also experience impaired communication during HRG surveys that use subbottom profilers and boomers, however, these surveys are expected to occur within the more coastal waters; therefore, exposure to HRG surveys would be limited. To preserve the saliency of their vocalizations, these whales may have to make one or more of the vocal adjustments discussed in this subsection. Because any reductions in the active space of whale vocalizations that result from sounds produced by seismic surveys would be temporary and episodic, vocal adjustments these whales would have to make would also be temporary.

MASKING. Marine mammals use acoustic signals for a variety of purposes, which differ among species, but include communication between individuals, navigation, foraging, reproduction, and learning about their environment (Dunlop *et al*. 2010, Erbé and Farmer 2000, Tyack 2000). Masking, or *auditory interference,* generally occurs when sounds in the environment are louder than and of a similar frequency to, auditory signals an animal is trying to receive. Masking, therefore, is a phenomenon that affects animals that are trying to receive acoustic information about their environment, including sounds from other members of their species, predators, prey, and sounds that allow them to orient in their environment. Masking these acoustic signals can disturb the behavior of individual animals, groups of animals, or entire populations (Box BR2 of Figure 3 illustrates the potential responses of animals to acoustic masking).

Richardson *et al*. (1995) argued that the maximum radius of influence of an industrial noise (including broadband low frequency sound transmission) on a marine mammal is the distance from the source to the point at which the noise can barely be heard. This range is determined by either the hearing sensitivity of the animal or the background noise level present. Industrial masking is most likely to affect some species' ability to detect communication calls and natural sounds (i.e., vocalizations from other members of its species, surf noise, prey noise, etc.; Richardson *et al*. 1995).

Masking affects primarily the weaker sound signals received from distant sound sources (Richardson et al 1995). The echolocation calls of toothed whales are subject to masking by high frequency sound. Blue and fin whales whose calls are capable of long-distance communication would also be subject to masking by the low frequencies emitted during seismic surveys. Human data indicate low frequency sound can mask high frequency sounds (i.e., upward masking). Studies on captive odontocetes by Au *et al*. (1974, 1985, 1993) indicate that some species may use various processes to reduce masking effects (e.g., adjustments in echolocation call intensity or frequency as a function of background noise conditions). These same coping mechanisms should be available to mysticetes as they would have to compete with the variability in natural background noise. The limits of these abilities, however, are unknown. There is also evidence that the directional hearing abilities of odontocetes are useful in reducing masking at the high frequencies these cetaceans use to echolocate, but not at the low-to-moderate frequencies they use to communication (Zaitseva *et al*. 1980).

For sea turtles and Atlantic sturgeon, hearing capabilities are centered in the low frequency range, as are the dominant frequencies of seismic pulses. However, it is not clear whether sea turtles or Atlantic sturgeon produce biologically significant sounds. If they do, and given the overlap of seismic frequencies and their hearing capabilities, it would not be unreasonable to anticipate some masking may result from the proposed seismic activities. Given the frequency overlap between seismic frequencies and hearing capabilities masking can occur, however, we do not have enough information to determine the extent of any masking should it occur.

Allostasis

Classic stress responses begin when an animal's central nervous system perceives a potential threat to its homeostasis. That perception triggers stress responses regardless of whether a stimulus actually threatens the animal; the mere perception of a threat is sufficient to trigger a stress response (Moberg 2000, Sapolsky *et al*. 2005, Seyle 1950). Once an animal's central nervous system perceives a threat, it mounts a biological response or defense that consists of a combination of the four general biological defense responses: behavioral responses, autonomic nervous system responses, neuroendocrine responses, or immune response.

In the case of many stressors, an animal's first and most economical (in terms of biotic costs) response is behavioral avoidance of the potential stressor or avoidance of continued exposure to a stressor (Box S1 of Figure 3). An animal's second line of defense to stressors involves the autonomic nervous system and the classical "fight or flight" response which includes the cardiovascular system, the gastrointestinal system, the exocrine glands, and the adrenal medulla to produce changes in heart rate, blood pressure, and gastrointestinal activity that humans commonly associate with "stress." These responses have a relatively short duration and may or may not have significant long-term effect on an animal's welfare.

An animal's third line of defense to stressors involves its neuroendocrine or sympathetic nervous systems; the system that has received the most study has been the hypothalmus-pituitary-adrenal system (also known as the HPA axis in mammals). Unlike stress responses associated with the autonomic nervous system, virtually all neuro-endocrine functions that are affected by stress – including immune competence, reproduction, metabolism, and behavior – are regulated by pituitary hormones. Stress-induced changes in the secretion of pituitary hormones have been implicated in failed reproduction (Moberg 1987, Rivier 1995, Box S2 of Figure 3) and altered metabolism (Elasser *et al*. 2000), reduced immune competence (Blecha 2000) and behavioral disturbance. Increases in the circulation of glucocorticosteroids (cortisol, corticosterone, and aldosterone in marine mammals; see Romano *et al*. 2004) have been equated with stress for many years.

The primary distinction between *stress* (which is adaptive and does not normally place an animal at risk) and *distress* is the biotic cost of the response. During a stress response, an animal uses glycogen stores that can be quickly replenished once the stress is alleviated. In such circumstances, the cost of the stress response would not pose a risk to the animal's welfare. However, when an animal does not have sufficient energy reserves to satisfy the energetic costs of a stress response, energy resources must be diverted from other biotic functions which impairs those functions that experience the diversion. For example, when mounting a stress response diverts energy away from growth in young animals, those animals may experience stunted growth. When mounting a stress response diverts energy from a fetus, an animal's reproductive success and its fitness will suffer. In these cases, the animals will have entered a pre-pathological or pathological state which is called "distress" (*sensu* Seyle 1950) or "allostatic loading" (*sensu* McEwen and Wingfield 2003). This pathological state will last until the animal replenishes its biotic reserves sufficient to restore normal function.

Relationships between these physiological mechanisms, animal behavior, and the costs of stress responses have also been documented fairly well through controlled experiment; because this physiology exists in every vertebrate that has been studied, it is not surprising that stress responses and their costs have been documented in both laboratory and free-living animals (for examples see, Holberton *et al*. 1996, Hood *et al*. 1998, Jessop *et al*. 2003, Krausman *et al*. 2004, Lankford *et al*. 2005, Reneerkens *et al*. 2002, Thompson and Hamer 2000). Although no information has been collected on the physiological responses of marine mammals upon exposure to anthropogenic sounds, studies of other marine animals and terrestrial animals would lead us to expect some marine mammals to experience physiological stress responses and, perhaps, physiological responses that would be classified as "distress" upon exposure to mid-frequency and low-frequency sounds.

For example, Jansen (1998) reported on the relationship between acoustic exposures and physiological responses that are indicative of stress responses in humans (for example, elevated respiration and increased heart rates). Jones (1998) reported on reductions in human performance when faced with acute, repetitive exposures to acoustic disturbance. Trimper *et al*. (1998) reported on the physiological stress responses of osprey to low-level aircraft noise while Krausman *et al*. (2004) reported on the auditory and physiology stress responses of endangered Sonoran pronghorn to military overflights. Smith *et al*. (2004a, 2004b) identified noise-induced physiological stress responses in hearing-specialist fish that accompanied short- (TTS) and long-term (PTS) hearing losses. Welch and Welch (1970), reported physiological and behavioral stress responses that accompanied damage to the inner ears of fish and several mammals.

Sea turtles exposed to seismic sound may possibly experience a physiological stress response, but available studies are inconclusive (DFO, 2004), and the magnitude or effects of any response remain largely unknown. Given evidence suggesting sea turtles likely avoid seismic sources, and those that are known to be exposed are not reported to have suffered a detectable physical injury, we assume that sea turtles exposed to seismic pulses during the proposed activities would not experience physical effects beyond a possible stress response.

There have been few studies on fishes (e.g., Smith et al., 2004b; Remage-Healey et al., 2006; Wysocki et al., 2006, 2007). There is some indication of physiological effects on fishes, such as a change in hormone levels and altered behavior, in some (Pickering, 1981; Smith et al., 2004a,b) but not all species tested to date (e.g., Wysocki et al., 2007). Sverdrup et al., 1994 found that Atlantic salmon subjected to up to 10 explosions to simulate seismic airguns released primary stress hormones, adrenaline and cortisol, as a biochemical response. There was no mortality. All experimental subjects returned to their normal physiological levels within 72 hrs of exposure.

Hearing is one of the primary senses marine mammals, sea turtles and fish use to gather information about their environment and to communicate with other members of their species. Although empirical information on the relationship between sensory impairment (TTS, PTS, and acoustic masking) on cetaceans remains limited, it seems reasonable to assume that reducing an animal's ability to gather information about its environment and to communicate with other members of its species would be stressful for animals that use hearing as their primary sensory mechanism. Therefore, we assume that acoustic exposures sufficient to trigger onset PTS or TTS would be accompanied by physiological stress responses because terrestrial animals exhibit those responses under similar conditions (NRC 2003). More importantly, marine mammals, sea turtles and fish might experience stress responses at received levels lower than those necessary to trigger onset TTS. Based on empirical studies of the time required to recover from stress responses (Moberg 2000), we also assume that stress responses are likely to persist beyond the time interval required for animals to recover from TTS and might result in pathological and pre-pathological states that would be as significant as behavioral responses to TTS.

Strandings

Available information indicates that marine mammals close to underwater detonations can be killed or severely injured, with auditory organs especially susceptible to injury (D.R. Ketten, Lien, & Todd, 1993; Ketten 1995 as cited in L-DEO, 2006). However, seismic airgun pulses are less energetic and have slower rise times, and there is no evidence available conclusively linking airguns to serious injury, death, or stranding even in the case of large airgun arrays (Gordon et al., 2003; L-DEO, 2006). Evidence implicating seismic airguns in the stranding of marine mammals does exist for two beaked whales in the Gulf of California (*Ziphius cavirostris* – not listed under the ESA nor currently considered a candidate for such listing), and possibly for four *Z. cavirostris* in the Galápagos Islands (Gentry, 2002; Gordon, et al., 2003). There appeared to be a temporal correlation between these events and seismic operations by the vessel *R/V Maurice Ewing* in the vicinity; however, a causal link could not be established for either event. Other strandings of beaked whales have also occurred, associated with military mid-frequency sonar transmissions – e.g., in the Bahamas, Canary Islands, and eastern Mediterranean Sea (Frantzis, 1998). These other strandings were associated with military mid-frequency sonar (generally 2-10 kHz and relatively narrow bandwidth), which differs from the sound produced by seismic arrays (broadband and below 1 kHz). However, evidence that sonar pulses can lead to physical damage or mortality (even if indirectly) (Fernández et al., 2005; Jepson et al., 2003; U.S. Department of Commerce & U.S. Department of the Navy, 2001) suggests that caution is still warranted when considering the effects on marine mammals from any high-intensity pulsed sound, such as from seismic airguns.

For sea turtles, no available information definitively links seismic airgun activities with any sea turtle mortalities (e.g., see 60 FR 21745; May 3, 1995) or strandings, although studies are limited. Anecdotal evidence from early 2004 indicates that more than 30 sea turtles stranded dead in Yucatán, Mexico (Jaszy & Horowitz, 2005). Guzman-Hernandez (pers. comm. in NMFS, 2006c) stated that one of the sea turtles had burst lungs, internal bleeding, and auditory damage. This event occurred during a time when seismic testing had been conducted nearby in very shallow waters. However, no definitive causal link is noted and seismic activity during the proposed study would be conducted with monitoring to allow an opportunity to keep leatherback sea turtles from exposures to the highest received levels.

Effects on Prey

In addition to the responses discussed above, surveys using active acoustics could have indirect, adverse effects on whales and sea turtles by reducing the abundance or availability of prey or changing the structure or composition of the fish community. Sperm whales and all species of sea turtles are likely to forage in some portion of the Action Area. The indirect effects of surveys could occur if fish or invertebrates experience lethal or sub-lethal damage, stress responses, or alterations in their behavior or distribution in response to acoustic energy produced by seismic surveys. Because fish and invertebrate species such as squid and jellyfish are pelagic prey for whales and sea turtles, such effects might have adverse consequences for individuals foraging in the Action Area.

Several studies have shown that short, sharp sounds can cause overt or subtle changes in fish behavior and distribution. Chapman and Hawkins (1969) tested the reactions of whiting (hake) in the field to an airgun. When the airgun was fired, the fish showed a sudden downward movement, changing their distribution from being dispersed between 25 m (80 ft) and 55 m (180 ft) depth, to forming a compact layer below 55 m (180 ft). Toward the end of an hour-long exposure to the airgun pulses the fish had habituated to the sound and risen back upward in the water column, despite the continued presence of sound pulses. However, when the airgun resumed firing after a dormant period, the fish exhibited another downward response. Pearson *et al.* (1992) conducted a controlled experiment to determine the effects of strong sound pulses on several species of rockfish off California. Exposing rockfish to an airgun with a source level of 223 dB re 1 μPa, the authors reported startle and alarm responses in these fish. Popper *et al.* (2005) report the occurrence of threshold shifts in some fish after exposure to airguns, with recovery in 24 hours. In other airgun experiments, catch-per-unit-effort (CPUE) of demersal fish was reported to decline when airgun pulses were emitted (Skalski *et al.* 1992; Dalen and Knutsen 1986, Dalen and Raknes 1985). Reductions in the catch may have resulted from a change in the behavior of the fish. Fish schools descended to near the bottom when the airgun was firing, and the fish may have changed their swimming and schooling behavior. Fish behavior returned to normal minutes after the sounds ceased. In the Barents Sea, abundance of cod and haddock measured acoustically was reduced by 44 percent within 9.2 km (5 nm) of an area where airguns operated (Engås *et al.* 1993). Actual catches declined by 50 percent throughout the trial area and 70 percent within the shooting area. This reduction in catch decreased with increasing distance until 30-33 km (16-18 nm), where catches were unchanged.

McCauley *et al.* (2003) also conducted an experiment on the effects of airgun sounds on fish. Several fish were exposed to an operating airgun, over 1.5 hours at 10-second intervals with received levels varying from less than 100 dB to over 160 dB re 1 μPa. The exposure resulted in apparent permanent, extensive damage to their sensory epithelia. Although this study demonstrates fish can be injured from repeated exposure to airgun sounds, fish in the wild are likely to move away from a seismic source and are not expected to be exposed in such a manner. Other studies of the behavioral responses of fish and fishing success to seismic sources report similar responses – e.g., see Dalen and Knutsen (1986), Turnpenny and Nedwell (1994), LaBella *et al.* (1996), Kenchington (1999), Santulli *et al.* (1999), Hirst and Rodhouse (2000), Thomson *et al.* (2001), Wardle *et al.* (2001), and Engås and Løkkeborg (2002). Egg and larval stages would not be able to escape such exposures, and Gausland (2000) reported that seismic signal levels of 230-240 dB re 1μPa pk-pk (or 16 dB lower: 214-224 dB rms) are sufficient for harm to occur to fish eggs and larvae (see also Kostyuchenko 1973).

A range of invertebrates are reported to be sensitive to low-frequency (10–150 Hz) hydroacoustic disturbances induced by sound waves or other sources – e.g., jelly fish, crustaceans, arrow worms, octopus, and squid (Western Australian Department of Industry and Resources 2002 in BOEM 2012a). This sensitivity overlaps the dominant frequency range of seismic pulses, indicating that invertebrates could likely perceive seismic activity (Western Australian Department of Industry and Resources 2002 in

BOEM 2012a). Available studies report responses to airgun shots as being limited to transient alarm responses such as tail-flicks (lobsters) or siphon closing (ascidians) (Western Australian Department of Industry and Resources 2002), although mortality of giant squid in the Bay of Biscay may possibly have been linked to seismic airgun activity in the area (Guerra *et al.* 2004). McCauley *et al*. (2000a; 2000b) examined the effect of marine seismic surveys on captive squid and cuttlefish and reported a strong startle response or directed movement away from airguns during sudden, nearby start-ups at received levels of 174 dB re 1 μPa mean squared pressure. Alarm responses in squid were detected during gradual ramp-up of airguns once levels exceeded 156-161 dB re 1 μPa mean squared pressure. Squid in these trials appeared to make use of the sound shadow measured near the water surface. These responses for captive squid suggest that behavioral changes and avoidance of operating airguns would likely occur. The authors concluded squid significantly alter their behavior at an estimated distance of 2–5 km (1.1–2.7 nm) from an approaching large seismic source. A more recent study by André et al (2011) exposed four species of squid and octopus in a tank to two hours of intense sounds. The authors reported tissue degeneration cause by the sounds. However, this study had substantial design and control problems and the results could have been attributed to those problems. Based on a review of airgun impacts on invertebrates by Christian and Bocking (2010), studies to date have not revealed any consistent evidence of serious pathological or physiological effects on invertebrates. With the seismic source vessel moving at speeds of about 4.5 kts (8.3 km/hr), the seismic vessel and its streamers would pass any given point within about an hour.

These studies indicate that seismic airgun activity has the potential to affect fish and invertebrates. Fish appear to exhibit startle responses and avoidance of seismic sources, recovering or habituating after a short time period. Squid also appear to exhibit alarm responses and avoidance of seismic sources. Limited data on the physiological effects of seismic sound on fish and invertebrates indicate these effects are short-term and most apparent after exposure at very close range. Although, disturbance of these prey species has the potential to negatively affect listed whales and sea turtles foraging in the Action Area, we expect such responses would have temporary effects on the feeding ability of whales and sea turtles in the immediate survey area.

Probable Responses of the Proposed Seismic and HRG Surveys

Thus far, this Opinion has identified the endangered and threatened species that might be exposed to sounds produced during seismic and HRG surveys within and adjacent to the Mid-Atlantic and South-Atlantic Planning Areas and the potential responses of those species given that exposure. The narratives that follow discuss the probable responses of these species to those surveys.

BLUE WHALE. Our analyses led us to reach the following conclusions about the potential stressors blue whales might be exposed to during seismic and HRG surveys within and adjacent to the Mid-Atlantic and South-Atlantic Planning Areas: blue whales are likely to respond to low- to mid-frequency sound sources associated with seismic surveys because those sounds are currently believed to occur within the best hearing sensitivity of blue whales. Blue whales have also been observed responding to sonar within the frequency ranges of 1-8 kHz (Melcón et al 2012).

While we recognize that animal hearing evolved separately from animal vocalizations and, as a result, it may be inappropriate to make inferences about an animal's hearing sensitivity from their vocalizations, we have no data on blue whale hearing so we assume that blue whale vocalizations are partially representative of their hearing sensitivities. Those vocalizations include a variety of sounds described as low frequency moans or long pulses in the 10-100 Hz band (Cummings and Thompson 1971; Edds 1982; Thompson and Friedl 1982; McDonald *et al.* 1995; Clark and Fristrup 1997; Rivers 1997). The most typical signals are very long, patterned sequences of tonal infrasonic sounds in the 15-40 Hz range. Ketten (1997) reports the frequencies of maximum energy between 12 and 18 Hz. Short sequences of rapid calls in the 30-90 Hz band are associated with animals in social groups (Clark personal observation and McDonald personal communication cited in Ketten 1997). The context for the 30-90 Hz calls suggests that they are used to communicate but do not appear to be related to reproduction. Blue whale moans within the frequency range of 12.5-200 Hz, with pulse duration up to 36 seconds, have been recorded off Chile (Cummings and Thompson 1971). The whale produced a short, 390 Hz pulse during the moan.

Nevertheless, we do not expect blue whales that find themselves within a sound field where received levels might be as high as 140 dB to devote attentional resources to that stimulus because those individuals are more likely to be focusing their attention on stimuli and environmental cues that are considerably closer, even if they were aware of the signal. Similarly, we would not expect blue whales that find themselves more than 20 km from the 5,400 in^3 airgun array, 6.4 km from the 90 in^3 array, 3.4 km from the chirp sub-bottom profiler or 8.2 km from the boomer or sparker to change their behavioral state (animals that are foraing, mating, or migrating will consintue to do so), despite being exposed to received levels ranging from 140 and 150 dB. These whales might engage in low-level avoidance behavior or short-term vigilance behavior. Blue whales that might occur between 0 and 21 km for the 5,400 in^3 airgun array, 0.186 km for the 90 in^3 airgun array 0.045 km (for the boomer and sparker) and 0.042 km (for the chirp subbottom profiler), are likely to change their behavioral state, although such a change is less likely if they are actively foraging. However, as we discussed previously, we do not assume that these blue whales would respond to only one sound source rather than all of the sounds produced by equipment employed during seismic and HRG surveys.

FIN WHALE. Our analyses led us to reach the following conclusions about the probable responses of fin whales exposed to seismic and HRG surveys within and adjacent to the Mid-Atlantic and South-Atlantic Planning Areas: like blue whales, fin whales are likely to respond to the low-frequency sound sources associated with seismic surveys because those sounds are currently believed to occur within the best hearing sensitivity of fin whales. While we recognize that animal hearing evolved separately from animal vocalizations and, as a result, it may be inappropriate to make inferences about an animal's hearing sensitivity from their vocalizations, we have no data on fin whale hearing so we assume that fin whale vocalizations are partially representative of their hearing sensitivities. Those vocalizations include a variety of sounds described as low frequency moans or long pulses in the 10-100 Hz band (Cummings and Thompson 1971; Edds 1982; Thompson and Friedl 1982; McDonald *et al.* 1995; Clark and Fristrup 1997; Rivers 1997). The most typical signals are very long, patterned sequences of tonal infrasonic sounds in the 15-40 Hz range. Ketten (1997) reported the frequencies of maximum energy lie between 12 and 18 Hz. Short sequences of rapid calls in the 30-90 Hz band are associated with animals in social groups (Clark personal observation and McDonald personal communication cited in Ketten 1997). The context for the 30-90 Hz calls suggests that fin whales use them to communicate but do not appear to be related to reproduction. Fin whale moans within the frequency range of 12.5-200 Hz, with pulse duration up to 36 seconds, have been recorded off Chile (Cummings and Thompson 1971). The whale produced a short, 390 Hz pulse during the moan.

Nevertheless, we do not expect fin whales that find themselves within a sound field where received levels might be as high as 140 dB to devote attentional resources to that stimulus because those individuals are more likely to be focusing their attention on stimuli and environmental cues that are considerably closer, even if they were aware of the signal. Similarly, we do not expect fin whales that find themselves more than 20 km from the 5,400 in^3 airgun array, 6.4 km from the 90 in^3 array, 3.4 km from the chirp sub-bottom profiler or 8.2 km from the boomer or sparker to change their behavioral state (animals that are foraing, mating, or migrating will consintue to do so), despite being exposed to received levels ranging from 140 and 150 dB. These whales might engage in low-level avoidance behavior or short-term vigilance behavior. Fin whales that might occur between 0 and 2.1 km for the 5,400 in^3 airgun array, 0.186 km for the 90 in^3 airgun array 0.045 km (for the boomer and sparker) and 0.042 km (for the chirp subbottom profiler), are likely to change their behavioral state, although such a change is less likely if they are actively foraging. However, as we discussed previously, we do not assume that these whales would respond to only one sound source rather than all of the sounds produced by equipment employed during seismic and HRG surveys.

HUMPBACK WHALE. Our analyses led us to reach the following conclusions about the probable responses of humpback whales exposed to seismic surveys within and adjacent to the Mid-Atlantic and South-Atlantic Planning Areas: as with blue and fin whales, humpback whales are also likely to respond to the low- to mid-frequency sound sources associated with seismic surveys because those sounds are currently believed to overlap with the best hearing sensitivity of these whales. As discussed in the *Status of the Species* narrative for humpback whales, these whales produce a wide variety of sounds. During the breeding season males sing long, complex songs, with frequencies in the 25-5000 Hz range and intensities as high as 181 dB (Payne 1970, Thompson *et al.* 1986, Winn *et al.* 1970). Source levels average 155 dB and range from 144 to 174 dB (Thompson *et al.* 1979). The songs appear to have an effective range of approximately 10 to 20 km. Animals in mating groups produce a variety of sounds (Silber 1986, Tyack 1981; Tyack and Whitehead 1983).

Humpback whales produce sounds less frequently in their summer feeding areas, which occur primarily north of the Action Area for this consultation. Feeding groups produce distinctive sounds ranging from 20 Hz to 2 kHz, with median durations of 0.2-0.8 seconds and source levels of 175-192 dB (Thompson *et al.* 1986). These sounds are attractive and appear to rally animals to the feeding activity (D'Vincent *et al.* 1985, Sharpe and Dill 1997). In summary, humpback whales produce at least three kinds of sounds:

1. Complex songs with components ranging from at least 20Hz – 4 kHz with estimated source levels from 144 – 174 dB; these are mostly sung by males on the breeding grounds (Payne 1970; Winn *et al.* 1970a; Richardson *et al.* 1995);
2. Social sounds in the breeding areas that extend from 50Hz – more than 10 kHz with most energy below 3 kHz (Tyack and Whitehead 1983, Richardson *et al.* 1995); and

3 Feeding area vocalizations that are less frequent, but tend to be 20Hz – 2 kHz with estimated sources levels in excess of 175 dB re 1 uPa-m (Thompson *et al.* 1986, Richardson *et al.* 1995). Sounds often associated with possible aggressive behavior by males (Silber 1986,Tyack 1983) are quite different from songs, extending from 50 Hz to 10 kHz (or higher), with most energy in components below 3 kHz. These sounds appear to have an effective range of up to 9 km (Tyack and Whitehead 1983).

More recently, Au *et al.* (2006) conducted field investigations of humpback whale songs led these investigators to conclude that humpback whales have an upper frequency limit reaching as high as 24 kHz. Based on this information, it is reasonable to assume that sounds produced by equipment employed during seismic and HRG surveys within and adjacent to the Mid-Atlantic and South-Atlantic Planning Areas are within the hearing and vocalization ranges of humpback whales.

Humpback whales have been known to react to low frequency industrial noises at estimated received levels of 115-124 dB (Malme *et al.* 1985), and to calls of other humpback whales at received levels as low as 102 dB (Frankel *et al.* 1995). Malme *et al.* (1985) found no clear response to playbacks of drill ship and oil production platform noises at received levels up to 116 dB re 1 μPa. Studies of reactions to airgun noises were inconclusive (Malme *et al.* 1985). Humpback whales on the breeding grounds did not stop singing in response to underwater explosions (Payne and McVay 1971). Humpback whales on feeding grounds did not alter short-term behavior or distribution in response to explosions with received levels of about 150dB re 1 μPa/Hz at 350Hz (Lien *et al.* 1993, Todd *et al.* 1996). However, at least two individuals were probably killed by the high-intensity, impulsed blasts and had extensive mechanical injuries in their ears (Ketten *et al.* 1993, Todd *et al.* 1996). The explosions may also have increased the number of humpback whales entangled in fishing nets (Todd *et al.* 1996). Frankel and Clark (1998) showed that breeding humpbacks showed only a slight statistical reaction to playbacks of 60 - 90 Hz sounds with a received level of up to 190 dB. Although these studies have demonstrated that humpback whales will exhibit short-term behavioral reactions to boat traffic and playbacks of industrial noise, the long-term effects of these disturbances on the individuals exposed to them are not known.

Based on this information, we do not expect humpback whales that find themselves within a sound field where received levels might be as high as 140 dB to devote attentional resources to that stimulus because those individuals are more likely to be focusing their attention on stimuli and environmental cues that are considerably closer, even if they were aware of the signal. Similarly, we do not expect humpback whales that find themselves more than 20 km from the 5,400 in^3 airgun array, 6.4 km from the 90 in^3 array, 3.4 km from the chirp sub-bottom profiler or 8.2 km from the boomer or sparker to change their behavioral state (animals that are foraing, mating, or migrating will consintue to do so), despite being exposed to received levels ranging from 140 and 150 dB. These whales might engage in low-level avoidance behavior or short-term vigilance behavior. Humpback whales that might occur between 0 and 2.1 km for the 5,400 in^3 airgun array, 0.186 km for the 90 in^3 airgun array 0.045 km (for the boomer and sparker) and 0.042 km (for the chirp subbottom profiler), are likely to change their behavioral state, although such a change is less likely if they are actively foraging. However, as we discussed previously, we do not assume that these whales would respond to only one sound source rather than all of the sounds produced by equipment employed during seismic and HRG surveys.

NORTH ATLANTIC RIGHT WHALE. Our analyses led us to reach the following conclusions about the probable responses of North Atlantic right whales exposed to seismic and HRG surveys within and adjacent to the Mid-Atlantic and South-Atlantic Planning Areas: as with the other whales we have discussed thus far, North Atlantic right whales are also likely to respond to the low-to mid-frequency sound sources associated with seismic surveys because those sounds are currently believed to occur within the best hearing sensitivity of right whales. The information available on right whales vocalizations suggests that right whales produce moans less than 400 Hz in frequency (Watkins and Schevill 1972; Thompson *et al.* 1979; Spero 1981), However, Nowacek *et al.* (2004) conducted controlled exposure experiments on North Atlantic right whales using ship noise, social sounds of con-specifics, and an alerting stimulus (frequency modulated tonal signals between 500 Hz and 4.5 kHz). Animals were tagged with acoustic sensors (D-tags) that simultaneously measured movement in three dimensions. Whales reacted strongly to alert signals at received levels of 133-148 dB SPL, mildly to conspecific signals, and not at all to ship sounds or actual vessels. Although the alert stimulus caused whales to immediately cease foraging behavior and swim rapidly to the surface, Nowacek *et al*. offer no information on whether the whales were probably responding to the low- or mid-frequency components of the signals.

Based on this information, we do not expect North Atlantic right whales that find themselves within a sound field where received levels might be as high as 140 dB to devote attentional resources to that stimulus because those individuals are more likely to be focusing their attention on stimuli and environmental cues that are considerably closer, even if they were aware of the signal. Similarly, we do not expect North Atlantic right whales that find themselves more than 20 km from the 5,400 in^3 airgun array, 6.4 km from the 90 in^3 array, 3.4 km from the chirp sub-bottom profiler or 8.2 km from the boomer or sparker to change their behavioral state (animals that are foraing, mating, or migrating will consintue to do so), despite being exposed to received levels ranging from 140 and 150 dB. These whales might engage in low-level avoidance behavior or short-term vigilance behavior. North Atlantic right whales that might occur between 0 and 2.1 km for the 5,400 in^3 airgun array, 0.186 km for the 90 in^3 airgun array 0.045 km (for the boomer and sparker) and 0.042 km (for the chirp subbottom profiler), are likely to change their behavioral state, although such a change is less likely if they are actively foraging. We do not believe that right whales are actively foraging in the ActionArea for this consultation as this area is a known migratory corridor for these whales. However, as we discussed previously, we do not assume that these whales would respond to only one sound source rather than all of the sounds produced by equipment employed during seismic and HRG surveys.

SEI WHALE. Our analyses led us to reach the following conclusions about the probable responses of sei whales exposed to seismic and HRG surveys within and adjacent to the Mid-Atlantic and South-Atlantic Planning Areas: Sei whales are also likely to respond to mid- to high-frequency sound sources associated with seismic surveys because of what we assume is their hearing sensitivity to low-frequency sounds. As discussed in the *Status of the Species* section of this opinion, we have no specific information on the sounds produced by sei whales or their sensitivity to sounds in their environment. Based on their anatomical and physiological similarities to both blue and fin whales, we assume that the hearing thresholds of sei whales will be similar as well and will be centered on low-frequencies in the 10-200 Hz. This information would lead us to conclude that, like blue and fin whales, sei whales exposed to these received levels of active mid-frequency sonar are not likely to respond if they are exposed to mid-frequency (1 kHz–10 kHz) sounds.

Based on this information, we do not expect sei whales that find themselves within a sound field where received levels might be as high as 140 dB to devote attentional resources to that stimulus because those individuals are more likely to be focusing their attention on stimuli and environmental cues that are considerably closer, even if they were aware of the signal. Similarly, we would not expect sei whales that find themselves more than 20 km from the 5,400 in^3 airgun array, 6.4 km from the 90 in^3 array, 3.4 km from the chirp sub-bottom profiler or 8.2 km from the boomer or sparker to change their behavioral state (animals that are foraing, mating, or migrating will consintue to do so), despite being exposed to received levels ranging from 140 and 150 dB. These whales might engage in low-level avoidance behavior or short-term vigilance behavior. Sei whales that might occur between 0 and 21 km for the 5,400 in^3 airgun array, 0.186 km for the 90 in^3 airgun array 0.045 km (for the boomer and sparker) and 0.042 km (for the chirp subbottom profiler), are likely to change their behavioral state, although such a change is less likely if they are actively foraging. However, as we discussed previously, we do not assume that these whales would respond to only one sound source rather than all of the sounds produced by equipment employed during seismic and HRG surveys.

SPERM WHALE. Our analyses led us to reach the following conclusions about the probable responses of sperm whales exposed to seismic and HRG surveys within and adjacent to the Mid-Atlantic and South-Atlantic Planning Areas: sperm whales are also likely to respond to sounds associated with the proposed HRG and seismic surveys. Although there is no published audiogram for sperm whales, sperm whales would be expected to have good, high frequency hearing because their inner ear resembles that of most dolphins, and appears tailored for ultrasonic (>20 kHz) reception (Ketten 1994). The only data on the hearing range of sperm whales are evoked potentials from a stranded neonate, which suggest that neonatal sperm whales respond to sounds from 2.5 to 60 kHz. Most of the energy of sperm whales clicks is concentrated at 2 to 4 kHz and 10 to 16 kHz, which overlaps with the mid-frequency sonar. Other studies indicate sperm whales' wide-band clicks contain energy between 0.1 and 20 kHz (Weilgart and Whitehead 1993, Goold and Jones 1995). Ridgway and Carder (2001) measured low-frequency, high amplitude clicks with peak frequencies at 500 Hz to 3 kHz from a neonate sperm whale. Nevertheless, several studies have demonstrated that sperm whales respond when exposed to sounds produced by seismic airguns, which suggests that sperm whales can hear spectral components of those sounds.

There is some evidence of disruptions of clicking and behavior from sonars (Goold 1999, Watkins and Scheville1975, Watkins *et al.* 1985), pingers (Watkins and Scheville 1975), the Heard Island Feasability Test (Bowles *et al.* 1994), and the Acoustic Thermometry of Ocean Climate (Costa *et al.*1998). Sperm whales have been observed to frequently stop echolocating in the presence of underwater pulses made by echosounders (Watkins and Scheville 1975). Goold (1999) reported six sperm whales that were driven through a narrow channel using ship noise, echosounder, and fishfinder emissions from a flotilla of 10 vessels. Watkins and Scheville (1975) showed that sperm whales interrupted click production in response to pinger (6 to 13 kHz) sounds. They also stopped vocalizing for brief periods when codas were being produced by other individuals, perhaps because they can hear better when not vocalizing themselves (Goold and Jones 1995).

Sperm whales have been reported to have reacted to mid-frequency active sonar, apparently produced by a submarine, by dispersing from social aggregations, moving away from the sound source, remaining relatively silent and becoming difficult to approach (Watkins *et al.* 1985). Captive bottlenose dolphins and a white whale exhibited changes in behavior when exposed to 1 sec pulsed sounds at frequencies similar to those emitted by multi-beam sonar that is used in geophysical surveys (Ridgway *et al.* 1997, Schlundt *et al.* 2000), and to shorter broadband pulsed signals (Finneran *et al.* 2000, 2002). Behavioral changes typically involved what appeared to be deliberate attempts to avoid the sound exposure or to avoid the location of the exposure site during subsequent tests (Schlundt *et al.* 2000, Finneran *et al.* 2002). Dolphins exposed to 1-sec intense tones exhibited short-term changes in behavior above received sound levels of 178 to 193 dB re 1 μPa rms and belugas did so at received levels of 180 to 196 dB and above. Received levels necessary to elicit such reactions to shorter pulses were higher (Finneran *et al.* 2000, 2002). Test animals sometimes vocalized after exposure to pulsed, mid-frequency sound from a watergun (Finneran *et al.* 2002). In some instances, animals exhibited aggressive behavior toward the test apparatus (Ridgway *et al.* 1997, Schlundt *et al.* 2000). The relevance of these data to free-ranging odontocetes is uncertain. In the wild, cetaceans some-times avoid sound sources well before they are exposed to the levels listed above, and reactions in the wild may be more subtle than those described by Ridgway *et al.* (1997) and Schlundt *et al.* (2000).

Other studies identify instances in which sperm whales did not respond to anthropogenic sounds. Sperm whales did not alter their vocal activity when exposed to levels of 173 dB re 1 μPa from impulsive sounds produced by 1 g TNT detonators (Madsen and Mohl 2000). Richardson *et al.* (1995) citing a personal communication with J. Gordon suggested that sperm whales in the Mediterranean Sea continued calling when exposed to frequent and strong military sonar signals. When Andre *et al.* (1997) exposed sperm whales to a variety of sounds to determine what sounds may be used to scare whales out of the path of vessels, sperm whales were observed to have startle reactions to 10 kHz pulses (180 db re 1 μPa at the source), but not to the other sources played to them.

Published reports identify instances in which sperm whales may have responded to an acoustic source and other instances in which they did not appear to respond behaviorally when exposed to seismic surveys. Mate *et al.* (1994) reported an opportunistic observation of the number of sperm whales to have decreased in an area after the start of airgun seismic testing. However, Davis *et al.* (2000) noted that sighting frequency did not differ significantly among the different acoustic levels examined in the northern Gulf of Mexico, contrary to what Mate *et al.* (1994) reported. In one DTAG deployment in the northern Gulf of Mexico on July 28, 2001, researchers documented that the tagged whale moved away from an operating seismic vessel once the seismic pulses were received at the tag at roughly 137 dB re 1 μPa (Johnson and Miller 2002). Sperm whales may also have responded to seismic airgun sounds by ceasing to call during some (but not all) times when seismic pulses were received from an airgun array >300 km away (Bowles *et al.* 1994).

A recent study offshore of northern Norway indicated that sperm whales continued to call when exposed to pulses from a distant seismic vessel. Received levels of the seismic pulses were up to 146 dB re 1 μPa peak-to-peak (Madsen *et al.* 2002). Similarly, a study conducted off Nova Scotia that analyzed recordings of sperm whale sounds at various distances from an active seismic program did not detect any obvious changes in the distribution or behavior of sperm whales (McCall Howard 1999). Recent data from vessel-based monitoring programs in United Kingdom waters suggest that sperm whales in that area may have exhibited some changes in behavior in the presence of operating seismic vessels (Stone 1997, 1998, 2000, 2001, 2003). However, the compilation and analysis of the data led the author to conclude that seismic surveys did not result in observable effects to sperm whales (Stone 2003). The results from these waters seem to show that some sperm whales tolerate seismic surveys.

Data from an experimental study of sperm whale reactions to seismic surveys in the Gulf of Mexico and a study of the movements of sperm whales with satellite-linked tags in relation to seismic surveys show that during two controlled exposure experiments in which sperm whales were exposed to seismic pulses at received levels up to 148 dB re 1 μPa over the octave band with the most energy, the whales did not avoid the vessel ((see A. Jochens et al., 2006; A. E. Jochens & Biggs, 2003; 2004). In a later publication concerning the experiments, Madsen *et al.* (2006) report that seven of eight tagged sperm whales continued to perform foraging dives throughout exposure to seismic airguns at levels up to 147 dB (rms) (the eighth whale remained at the surface during exposure). Although the sample size is small (4 whales in 2 experiments), the results are consistent with those off northern Norway.

These studies suggest that the behavioral responses of sperm whales to anthropogenic sounds are highly variable, but do not appear to result in the death or injury of individual whales or result in reductions in the fitness of individuals involved. Responses of sperm whales to anthropogenic sounds probably depend on the age and sex of animals being exposed, as well as other factors. There is evidence that many individual sperm whales are likely to respond to seismic surveys while other individuals will not.

Sea Turtles

Our analyses led us to reach the following conclusions about the probable responses of sea turtles exposed to seismic and HRG surveys within and adjacent to the Mid-Atlantic and South-Atlantic Planning Areas:

The available information on sea turtle hearing thresholds is limited; however, we assume that sea turtles have a practical hearing range from about 50 Hz upwards to a practical limit of 1,600 Hz where sensitivity sharply declines. Given the frequencies emitted by airguns and electromechanical equipment, the low frequency components emitted by airguns (50 Hz to 2 kHz), boomers (200 Hz to 16 kHz) and sparkers (50-200 Hz) and possibly the chirp depth sounder would be audible to sea turtles. Continuous sounds from drilling equipment would also be audible to sea turtles. Sea turtles are not expected to hear the chirp subbottom profilers at any frequency (3.5, 12 and 200 kHz), the side-scan sonars (100 kHz to 900 kHz), the single or multibeam depth sounders (240 kHz) or the dipole used during CSEM surveys (<2 Hz).

Avoidance of seismic sources by sea turtles is likely, but some turtles may not avoid the source vessel and may be exposed to seismic sound at levels of concern. Based on available information on captive turtles, avoidance may begin at received levels above 166 dB re 1 μPa. Avoidance behavior may shorten the exposure period, and the ramping-up of airguns during the proposed activities would provide opportunity for avoidance by sea turtles, thereby minimizing exposure to received levels of concern. Avoidance or any disruptions in sea turtle behavior are expected to be temporary and are not expected to cause any injury. For those turtles that might be exposed to seismic pulses at levels above 166 dB re 1 μPa during the proposed activities, we expect this could result in a stress response that rises to the level of harassment.

Atlantic Sturgeon

Our analyses led us to reach the following conclusions about the probable responses of sea turtles exposed to seismic and HRG surveys within and adjacent to the Mid-Atlantic and South-Atlantic Planning Areas:

Sturgeon are categorized as fishes that detect sounds from below 50 Hz to perhaps 800-1,000 Hz (though several probably only detect sounds to 600-800 Hz). These fishes have a swim bladder but no known structures in the auditory system that would enhance hearing, and sensitivity (lowest sound detectable at any frequency) is not very great. Sounds would have to be more intense to be detected compared to fishes with swim bladders that enhance hearing. Sturgeon can detect both particle motion and pressure.

Atlantic sturgeon are expected to primarily be exposed to HRG surveys conducted under the renewable energy and the marine minerals programs. Seismic survey activities are conducted in the more offshore waters but may still be heard at lower intensities. Given the frequencies emitted by airguns and electromechanical equipment, the low frequency components emitted by airguns (50 Hz to 2 kHz), boomers (200 Hz to 16 kHz) and sparkers (50-200 Hz) would be audible to Atlantic sturgeon. Continuous sounds from drilling equipment would also be audible but because drilling activities are further offshore than Atlantic sturgeon known occurrence, we do not expect them to be exposed to these sounds. Atlantic sturgeon are not expected to hear the chirp subbottom profilers at any frequency (3.5, 12 and 200 kHz), the side-scan sonars (100 kHz and 400 kHz), the single or multibeam depth sounders (240 kHz) or the dipole used during CSEM surveys (<2 Hz).

Atlantic sturgeon are expected to avoid seismic or HRG sources, but some may not avoid the source vessel and may be exposed to seismic sound at levels of concern. Avoidance behavior may shorten the exposure period. Avoidance or any disruptions in Atlantic sturgeon behavior are expected to be temporary and are not expected to cause any injury. Stress responses that rise to the level of harassment are not expected.

Effects Resulting from Interactions of the Potential Stressors

Several organizations have argued that several of our previous biological opinions on the U.S. Navy's use of active sonar failed to consider the "cumulative impact" (in the NEPA sense of the term) of active sonar on the ocean environment and its organisms, particularly endangered and threatened species and critical habitat that has been designated for them. In this Opinion, we have explained how biological opinions consider "cumulative impacts" (in the NEPA sense of the term; see Approach to the Assessment for a complete treatment of this issue). There is a nuance to the idea of "cumulative impacts," however, that we have chosen to address explicitly in this Opinion: potential interactions between stressors associated with G&G activities and other physical, chemical, and biotic stressors that pre-exist in the environment.

Exposing living organisms to individual stressors or a suite of stressors that are associated with a specific action may be insignificant or minor when considered in isolation, but may have significant adverse consequences when they are added to other stressors, operate synergistically in combination with other stressors, or magnify or multiply the effects of other stressors. Further, the effects of life events, natural phenomena, and anthropogenic phenomena on an individual's performance will depend on the individual's phenotypic state when the individual is exposed to these phenomena. Disease, dietary stress, body burden of toxic chemicals, energetic stress, percentage body fat, age, reproductive state, and social position, among many other phenomena can "accumulate" to have substantial influence on an organism's response to subsequent exposure to a stressor. That is, exposing animals to individual stressors associated with a specific action can interact with the animal's prior condition (can "accumulate" and have additive, synergistic, magnifying, and multiplicative effect) and produce significant, adverse consequences that would not occur if the animal's prior condition had been different.

An illustrative example of how a combination of stressors interact was provided by Relyea (2000, 2009) who demonstrated that exposing several different amphibians to a combination of pesticides and chemical cues of natural predators, which induced stress, increased the mortality rates of the amphibians (see also Sih, Bell, & Kerby, 2004). For some species, exposing the amphibians to the combination of stressors produced mortality rates that were twice as high as the mortality rates associated with each individual stressor. This section considers the evidence available to determine if interactions associated with seismic and HRG survey activities are likely to produce responses we have not considered already or if interactions are likely to increase the severity —and, therefore, the potential consequences — of the responses we have already considered.

The seismic and HRG survey activities BOEM proposes to authorize within and adjacent to the mid- and South Atlantic Planning Areas from 2013-2020 would add approximately 90,000 hours of seismic and HRG transmissions combined (assuming a vessel speed of 4.5kts (8.3km/hr), under the oil and gas program, approximately 75, 000 hours of HRG tranismissions under the renewable energy program and approximately 1,450 hours of HRG transmissions under the marine minerals program) to ambient oceanic noise levels, which, in turn, could have cumulative impacts on the ocean environment in the Action Area and any endangered or threatened species that occur in that area. The behavioral responses of any endangered whales that occur in those sound fields are likely to result from their response to low and mid-frequencies associated with seismic and electromechaniceal equipment, per se, as well as other salient cues in their environment, including their perception of their distance from a sound source and their perception of whether the sound source is approaching them, moving parallel to them, or moving away from them. The behavioral response of endangered whales that are likely to be exposed to seismic and electromechanical equipment during G&G activities considered in this Opinion would also be shaped by their reproductive condition, their state of health, and their prior experience.

With the possible exception of hatchling sea turtles or new-born calves, the prior experience of any individual endangered whales or listed sea turtles that occur off the Atlantic coast of the United States would include regular exposure to a wide variety of natural and anthropogenic stressors produced by commercial fisheries in Federal and State waters, vessel traffic, whale-watch vessels and researchers, and human occupation of coastal areas. Whales that occur in the Action Area are almost certain to have been exposed to ship traffic entering and leaving major ports like Baltimore, Virginia Beach, and Jacksonville. As we discussed in the *Environmental Baseline* section of this Opinion, these whales will also have been exposed to the continuous, low-frequency sounds produced by commercial vessels, may have interacted with commercial fishing gear, may have prior experience with mid-frequency active sonar from surface vessels, sonobuoys, or submarines in other geographic areas or during previous U.S. Navy training exercises, and given their long life spans may have prior experience with oil and gas drilling activities within the Action Area between 1975 and 1984.

Any individual humpback or sperm whales that have high body burdens of some antibiotics and other prescription drugs (which are introduced to marine environments in sewage discharges and freshwater run-off), organic solvents, metals, and chemical asphyxiants may be more susceptible to noise-induced loss of hearing sensitivity (Fechter, Chen, Rao, & Larabee, 2000). Nevertheless, none of the information available suggests that the responses of whales to the cumulative effects of these stressors or their prior experience with these stressors is likely to produce consequences other than those we have already considered in this Opinion although it is important to note that this statement probably results from the limits in the information available more than the absence of interactive effects.

Sea turtles that occur in the Action Area are almost certain to have been exposed to ship traffic entering and leaving ports like Boston, New York, Baltimore, Virginia Beach, and Jacksonville (if they arrive from the north) or Port Everglades and Fort Lauderdale (if they arrive from the south). As we discussed in the *Environmental Baseline* section of this Opinion, these sea turtles will also have been exposed to the continuous, low-frequency sounds produced by commercial vessels, are likely to have been captured at least once in commercial fishing gear (given that almost twice the estimated population of loggerhead sea turtles has been estimated to be captured in shrimp trawls each year for the past five years, every loggerhead sea turtle in the western Atlantic has had some probability of being captured in these fisheries), and may have prior experience with surface vessels, sonobuoys, or submarines in other geographic areas or during U.S. Navy exercises within the Action Area. Nevertheless, none of the information available suggests that the responses of sea turtles to these stressors or their prior experience with these stressors is likely to produce consequences other than those we have already considered in this Opinion, although it is important to note that this statement probably results from the limits in the information available more than the absence of interactive effects.

Similarily, Atlantic sturgeon that occur in the Action Area are almost certain to have been exposed to ship traffic entering and leaving ports like Boston, New York, Baltimore and Virginia Beach. As we discussed in the *Environmental Baseline* section of this Opinion, Atlantic sturgeon will also have been exposed to the continuous, low-frequency sounds produced by commercial vessels and dredging operations and are likely to have been captured in commercial fishing gear. Nevertheless, none of the information available suggests that the responses of Atlantic sturgeon to these stressors or their prior experience with these stressors is likely to produce consequences other than those we have already considered in this Opinion, although it is important to note that this statement probably results from the limits in the information available more than the absence of interactive effects.

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.

During this consultation, NMFS searched for information on future State, tribal, local, or private actions that were reasonably certain to occur in the action area. Most of the action area includes federal military reserves or is outside of territorial waters of the United States of America, which would preclude the possibility of future state, tribal, or local action that would not require some form of federal funding or authorization. NMFS conducted electronic searches of business journals, trade journals, and newspapers using *First Search*, Google, and other electronic search engines. Those searches produced no evidence of future private action in the action area that would not require federal authorization or funding and is reasonably certain to occur. As a result, NMFS is not aware of any actions of this kind that are likely to occur in the action area during the foreseeable future.

Integration and Synthesis of Effects

In the *Assessment Approach* section of this Opinion, we stated that we measure risks to individuals of endangered or threatened species using changes in the individual's "fitness" or the individual's growth, survival, annual reproductive success, and lifetime reproductive success. When we do not expect listed plants or animals exposed to an action's effects 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 (Anderson 2000, Mills and Beatty 1979, Brandon 1978, Stearns 1977, 1992). As a result, if we conclude that listed species are *not* likely to experience reductions in their fitness, we would conclude our assessment. If, however, we conclude that listed species are likely to experience reductions in their fitness, we assess the potential effects of the action on the viability of the population or populations' those individuals represent. At the population level, we would generally assume that an action that increased a population's probability of becoming extinct would place an endangered or threatened species at greater risk of extinction because species become extinct as a result of the extinction of the populations that comprise them.

The following discussions separately summarize the probable risks seismic and HRG activities pose to threatened and endangered species that are likely to be exposed to those transmissions. These summaries integrate the exposure profiles presented previously with the results of our response analyses for each of the actions considered in this Opinion.

Marine Mammals

BLUE WHALES. We did not estimate the number of blue whales that might be exposed to vessel traffic independent of the number of individuals that might be exposed to seismic and HRG surveys because the data we would have needed to support those analyses were not available.

Blue whales would primarily be exposed to the more offshore seismic and HRG surveys under the oil and gas program; not the more coastal surveys such as those under the renewable energy and marine mineral resource surveys. Blue whales migrate out of the Action Area to the more northerly waters outside of the Action Area to feed. Based on the BOEMs models, each year we would expect that numbers of blue whales might be exposed to seismic and HRG surveys and be "taken" as a result of that exposure (see Tables 13 and 14 later in this section). Most of these exposures are expected in the mid-Atlantic Planning Area.

Blue whales appear to be able to hear mid-frequency (1 kHz–8 kHz) sounds as well as low frequency sounds. Blue whales vocalizations include a variety of sounds described as low frequency moans or long pulses in the 10-100 Hz band (Cummings and Thompson 1971; Edds 1982; Thompson and Friedl 1982; McDonald *et al.* 1995; Clark and Fristrup 1997; Rivers 1997). The most typical signals are very long, patterned sequences of tonal infrasonic sounds in the 15-40 Hz range. Ketten (1997) reports the frequencies of maximum energy between 12 and 18 Hz. Short sequences of rapid calls in the 30-90 Hz band are associated with animals in social groups (Clark personal observation and McDonald personal communication cited in Ketten 1997). The context for the 30-90 Hz calls suggests that they are used to communicate but do not appear to be related to reproduction. Blue whale moans within the frequency range of 12.5-200 Hz, with pulse duration up to 36 seconds, have been recorded off Chile (Cummings and Thompson 1971). The whale produced a short, 390 Hz pulse during the moan. Based on this information blue whales exposed to received levels of active mid-frequency sonar are not likely to respond physiologically or behaviorally to sounds in this frequency range.

Blue whales in the action area seem likely to respond to the ship traffic associated with seismic and HRG survey activities in ways that approximate their responses to whale watch vessels. As discussed in the *Environmental Baseline* section of these Opinions, those responses are likely to depend on the distance of a whale from a vessel, vessel speed, vessel direction, vessel noise, and the number of vessels involved in a particular maneuver. Blue whales seem most likely to try to avoid being exposed to the activities and their avoidance response is likely to increase as survey activities increase. We do not have the information necessary to determine which of the many sounds associated with a seismic or HRG survey is likely to trigger avoidance behavior in blue whales (for example, engine noise, airguns, electromechanical equipment, or some combination of these) or whether blue whales would avoid being exposed to specific received levels, the entire sound field associated with a survey, or the general area in which a survey would occur. However, blue whales are not likely to respond to the side-scan sonar, single and multibeam echo sounders or the 200 kHz chirp subbottom profiler because they are not likely to hear those transmissions.

Individual blue whales might not respond to the vessels, while in other circumstances, whales are likely to change their surface times, swimming speed, swimming angle or direction, respiration rates, dive times, feeding behavior, and social interactions (Amaral and Carlson 2005; Au and Green 2000, Cockeron 1995, Erbe 2002, Félix 2001, Magalhães *et al.* 2002, Richter *et al.* 2003, Scheidat *et al.* 2004, Simmonds 2005, Watkins 1986, Williams *et al.* 2002). Some of these whales might experience physiological stress (but not "distress") responses to seismic surveys.

Several authors have established that long-term and intense disturbance stimuli can cause population declines by reducing the body condition of individuals that have been disturbed, followed by reduced reproductive success, reduced survival, or both (Daan *et al*. 1996, Madsen 1994, White 1983). For example, Madsen (1994) reported that pink-footed geese (*Anser brachyrhynchus*) in undisturbed habitat gained body mass and had about a 46% reproductive success compared with geese in disturbed habitat (being consistently scared off the fields on which they were foraging) which did not gain mass and has a 17% reproductive success. Similar reductions in reproductive success have been reported for mule deer (*Odocoileus hemionus*) disturbed by all-terrain vehicles (Yarmoloy *et al*. 1988), caribou disturbed by seismic exploration blasts (Bradshaw *et al*. 1998), caribou disturbed by low-elevation military jet-fights (Luick *et al*. 1996), and caribou disturbed by low-elevation jet flights (Harrington and Veitch 1992). Similarly, a study of elk (*Cervus elaphus*) that were disturbed experimentally by pedestrians concluded that the ratio of young to mothers was inversely related to disturbance rate (Phillips and Alldredge 2000). The primary mechanism by which increased vigilance and disturbance appear to affect the fitness of individual animals is by disrupting an animal's time budget and, as a result, reducing the time they might spend foraging and resting (which increases an animal's activity rate and energy demand). For example, a study of grizzly bears (*Ursus horribilis*) reported that bears disturbed by hikers reduced their energy intake by an average of 12 kcal/min (50.2 x 103kJ/min), and spent energy fleeing or acting aggressively toward hikers (White *et al*. 1999).

Despite seismic survey concentrated off the coast of Southern California, since 1968 and a total of 157,420 km of 2D surveys between 1968 and 2002, blue whales started being recorded in California waters in the 1970s. Although blue whales distribution has shifted, prey abundance shifts are believed to be the cause (Caretta et al 2009b in Thomsen et al 2011).

Because primary blue whale foraging habitat is in more northerly waters outside of the Action Area, we do not expect costly time and energy budget reductions due to lost foraging opportunities where the fitness of blue whales is reduced. Blue whales that are resting at the start of seismic or HRG survey activities will try to avoid the sound sources, if the sources are close enough to be perceived as a potential threat. While some changes in behavioral state, such as resting to traveling, incurs some energetic cost, these changes are not likely to alter the physiology, behavioral ecology and social dynamics of individual whales in ways or to a degree that would reduce the fitness of these whales.

Based on the evidence available, we conclude that the G&G activities BOEM and BSEE propose in the mid- and South Atlantic Planning Areas from 2013 June 2020 is not likely to adversely affect the population dynamics, behavioral ecology, and social dynamics of individual blue whales in ways or to a degree that would reduce their fitness. As we discussed in the *Approach to the Assessment* section of this Opinion, an action that is not likely to reduce the fitness of individual whales would not be likely to reduce the viability of the populations those individual whales represent (that is, we would not expect reductions in the reproduction, numbers, or distribution of those populations). As a result, we conclude that the G&G activities BOEM and BSEE propose from 2013 to 2020 would not appreciably reduce the blue whales' likelihood of surviving and recovering in the wild.

FIN WHALES. We did not estimate the number of fin whales that might be exposed to vessel traffic independent of the number of individuals that might be exposed to seismic and HRG surveys because the data we would have needed to support those analyses were not available.

Based on the BOEM's models, each year we would expect that numbers of fin whales might be exposed to seismic and HRG surveys and be "taken" as a result of that exposure (see Tables 13 and 14). Most of these exposures are expected in the mid-Atlantic Planning Area. Fin whales would primarily be exposed in the more offshore areas of the mid-Atlantic Planning Area mostly in the fall, winter and spring. During the summer months, fin whales tend to congregate in feeding areas between 41°20'N and 51°00'N, to northern Areas outside of the Action Area to feed.

As discussed in the *Status of the Species* section of these Opinions, fin whales produce a variety of low-frequency sounds in the 10-200 Hz band (Watkins 1981; Watkins *et al.* 1987a; Edds 1988; Thompson *et al.* 1992). The most typical signals are long, patterned sequences of short duration (0.5-2s) infrasonic pulses in the 18-35 Hz range (Patterson and Hamilton 1964). Estimated source levels are as high as 190 dB (Patterson and Hamilton 1964; Watkins *et al.* 1987a; Thompson *et al.* 1992; McDonald *et al.* 1995). 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 (Clark and Charif 1998). Short sequences of rapid pulses in the 20-70 Hz band are associated with animals in social groups (McDonald *et al.* 1995). Each pulse lasts on the order of one second and contains twenty cycles (Tyack 1999). This information would lead us to conclude that fin whales exposed to these received levels of active mid-frequency sonar are not likely to respond physiologically or behaviorally.

Fin whales in the action area seem likely to respond to the ship traffic associated with seismic and HRG survey activities in ways that approximate their responses to whale watch vessels. As discussed in the *Environmental Baseline* section of these Opinions, those responses are likely to depend on the distance of a whale from a vessel, vessel speed, vessel direction, vessel noise, and the number of vessels involved in a particular maneuver. Fin whales seem most likely to try to avoid being exposed to the activities and their avoidance response is likely to increase as survey activities increase. We do not have the information necessary to determine which of the many sounds associated with a seismic or HRG survey is likely to trigger avoidance behavior in fin whales (for example, engine noise, airguns, electromechanical equipment, or some combination of these) or whether fin whales would avoid being exposed to specific received levels, the entire sound field associated with a survey, or the general area in which a survey would occur. However, fin whales are not likely to respond to the side-scan sonar, single and multibeam echo sounders or the 200 kHz chirp subbottom profiler because they are not likely to hear those transmissions.

Particular whales might not respond to the vessels, while in other circumstances, fin whales are likely to change their vocalizations, surface time, swimming speed, swimming angle or direction, respiration rates, dive times, feeding behavior, and social interactions (Amaral and Carlson 2005; Au and Green 2000, Cockeron 1995, Erbe 2002, Félix 2001, Magalhães *et al.* 2002, Richter *et al.* 2003, Scheidat *et al.* 2004, Simmonds 2005, Watkins 1986, Williams *et al.* 2002). Some of these whales might experience physiological stress (but not "distress"). Some of these whales might experience physiological stress (but not "distress") responses to seismic surveys and HRG surveys.

Several authors have established that long-term and intense disturbance stimuli can cause population declines by reducing the body condition of individuals that have been disturbed, followed by reduced reproductive success, reduced survival, or both (Daan *et al*. 1996, Madsen 1994, White 1983). For example, Madsen (1994) reported that pink-footed geese (*Anser brachyrhynchus*) in undisturbed habitat gained body mass and had about a 46% reproductive success compared with geese in disturbed habitat (being consistently scared off the fields on which they were foraging) which did not gain mass and has a 17% reproductive success. Similar reductions in reproductive success have been reported for mule deer (*Odocoileus hemionus*) disturbed by all-terrain vehicles (Yarmoloy *et al*. 1988), caribou disturbed by seismic exploration blasts (Bradshaw *et al*. 1998), caribou disturbed by low-elevation military jet-fights (Luick *et al*. 1996), and caribou disturbed by low-elevation jet flights (Harrington and Veitch 1992). Similarly, a study of elk (*Cervus elaphus*) that were disturbed experimentally by pedestrians concluded that the ratio of young to mothers was inversely related to disturbance rate (Phillips and Alldredge 2000). The primary mechanism by which increased vigilance and disturbance appear to affect the fitness of individual animals is by disrupting an animal's time budget and, as a result, reducing the time they might spend foraging and resting (which increases an animal's activity rate and energy demand). For example, a study of grizzly bears (*Ursus horribilis*) reported that bears disturbed by hikers reduced their energy intake by an average of 12 kcal/min (50.2 x 103kJ/min), and spent energy fleeing or acting aggressively toward hikers (White *et al*. 1999).

Despite seismic surveys concentrated off the coast of Southern California, since 1968 and a total of 157,420 km of 2D surveys between 1968 and 2002, fin whale abundance indicate that numbers in that area have remained the same since the 1990s. No definitive statement can be made for fin whale abundance trends because of uncertainty in the assessments (Caretta et al 2009b in Thomsen et al 2011). Bowhead whales in the Arctic have been exposed to seismic surveys due to oil and gas exploration, oil and gas development and production and decommissioning activities since the late 1960s. Despite this exposure, the western Arctic stock of bowhead whales has been increasing at a rate of approximately 3.4 percent per year (George et al 2004).

Because primary fin whale foraging habitat is in more northerly waters outside of the Action Area, we do not expect costly time and energy budget reductions due to lost foraging opportunities where the fitness of fin whales is reduced. Fin whales that are resting at the start of seismic or HRG survey activities will try to avoid the sound sources, if the sources are close enough to be perceived as a potential threat. While some changes in behavioral state, such as resting to traveling, incurs some energetic cost, these changes are not likely to alter the physiology, behavioral ecology and social dynamics of individual whales in ways or to a degree that would reduce the fitness of these whales. As a result, we conclude that the G&G activities BOEM and BSEE proposes from 2013 to 2020 would not appreciably reduce the fin whales' likelihood of surviving and recovering in the wild.

HUMPBACK WHALES. We did not estimate the number of humpback whales that might be exposed to vessel traffic independent of the number of individuals that might be exposed to seismic and HRG surveys because the data we would have needed to support those analyses were not available.

Humpback whales would be exposed to seismic and HRG surveys primarily during the winter months and in the mid-Atlantic Planning Area rather than the South Atlantic Planning Area. Juvenile humpbacks tend to congregate in the mid-Atlantic region during winter months. Adults migrate through the mid- and South Atlantic Planning area to their winter breeding/calving area in the West Indies and back in the spring to the more northerly waters outside of the Action Area to feed. Based on the BOEM's models, each year we would expect that numbers of humpback whales might be exposed to seismic and HRG surveys and be "taken" as a result of that exposure (see Tables 13 and 14 later in this section).

Humpback whales produce a wide variety of sounds. During the breeding season males sing long, complex songs, with frequencies in the 25-5000 Hz range and intensities as high as 181 dB (Payne 1970; Winn *et al.* 1970a; Thompson *et al.* 1986). Source levels average 155 dB and range from 144 to 174 dB (Thompson *et al.* 1979). The songs appear to have an effective range of approximately 10 to 20 km. Animals in mating groups produce a variety of sounds (Tyack 1981; Tyack and Whitehead 1983, Silber 1986).

Humpback whales produce sounds less frequently in their summer feeding areas. Feeding groups produce distinctive sounds ranging from 20 Hz to 2 kHz, with median durations of 0.2-0.8 sec and source levels of 175-192 dB (Thompson *et al.* 1986). These sounds are attractive and appear to rally animals to the feeding activity (D'Vincent *et al.* 1985; Sharpe and Dill 1997). In summary, humpback whales produce at least three kinds of sounds:

> 1. Complex songs with components ranging from at least 20Hz – 4 kHz with estimated source levels from 144 – 174 dB; these are mostly sung by males on the breeding grounds (Payne 1970; Winn *et al.* 1970a; Richardson *et al.* 1995)
> 2. Social sounds in the breeding areas that extend from 50Hz – more than 10 kHz with most energy below 3kHz (Tyack and Whitehead 1983, Richardson *et al.* 1995); and
> 3. Feeding area vocalizations that are less frequent, but tend to be 20Hz – 2 kHz with estimated sources levels in excess of 175 dB re 1 uPa-m (Thompson *et al.* 1986; Richardson *et al.* 1995).
>
> Sounds often associated with possible aggressive behavior by males (Tyack 1983; Silber 1986) are quite different from songs, extending from 50 Hz to 10 kHz (or higher), with most energy in components below 3 kHz. These sounds appear to have an effective range of up to 9 km (Tyack and Whitehead 1983).

More recently, Au *et al.* (2006) conducted field investigations of humpback whale songs that led these investigators to conclude that humpback whales have an upper frequency limit reaching as high as 24 kHz. Based on this information, it is reasonable to assume that the active mid-frequency sonar the U.S. Navy would employ during the proposed active sonar training activities are within the hearing and vocalization ranges of humpback whales. There is limited information on how humpback whales are likely to respond upon being exposed to mid-frequency active sonar (most of the information available addresses their probable responses to low-frequency active sonar or impulsive sound sources). Humpback whales responded to sonar in the 3.1–3.6 kHz by swimming away from the sound source or by increasing their velocity (Maybaum 1990, 1993). The frequency or duration of their dives or the rate of underwater vocalizations, however, did not change.

Humpback whales have been known to react to low frequency industrial noises at estimated received levels of 115- 124 dB (Malme *et al.* 1985), and to conspecific calls at received levels as low as 102 dB (Frankel *et al.* 1995). Malme *et al.* (1985) found no clear response to playbacks of drill ship and oil production platform noises at received levels up to 116 dB re 1 Pa. Studies of reactions to airgun noises were inconclusive (Malme *et al.* 1985). Humpback whales on the breeding grounds did not stop singing in response to underwater explosions (Payne and McVay 1971). Humpback whales on feeding grounds did not alter short-term behavior or distribution in response to explosions with received levels of about 150dB re 1 Pa/Hz at 350Hz (Lien *et al.* 1993; Todd *et al.* 1996). Frankel and Clark (1998) showed that breeding humpbacks showed only a slight statistical reaction to playbacks of 60 - 90 Hz sounds with a received level of up to 190 dB. Although these studies have demonstrated that humpback whales will exhibit short-term behavioral reactions to boat traffic and playbacks of industrial noise, the long-term effects of these disturbances on the individuals exposed to them are not known.

Because their hearing range appears to overlap with the frequency range of low and mid-frequencies, we assume that some of the humpback whales that are exposed to seismic and HRG surveys might experience physiological stress responses as a result of their exposure. Acoustic masking and impairment of acoustic communication may also occur, although seismic and HRG surveys are a considerable distance from humpback foraging grounds. We expect humpback whale to adjust their vocalizations to compensate for the reduced communication space.

The evidence available suggests that humpback whales are likely to detect low and mid-frequency seismic and HRG transmissions. In most circumstances, humpback whales are likely to try to avoid that exposure or are likely to avoid specific areas. As a result, we do not expect these disruptions to reduce the fitness (reproductive success or longevity) of any individual animal or to result in physiological stress responses that rise to the level of distress.

The strongest evidence of the probable impact of the *Environmental Baseline* on humpback whales consists of the estimated growth rate of the humpback whale population in the Atlantic Ocean. Despite small numbers that are entangled in fishing gear in the action area, this increase in the number of humpback whales suggests that the stress regime these whales are exposed to in the Atlantic Ocean have not prevented these whales from increasing their numbers in the action area. As discussed in the *Environmental Baseline* section of this Opinion, humpback whales have been exposed to active sonar training activities along the Atlantic Coast of the United States and in the Gulf of Mexico, including vessel traffic, aircraft traffic, active sonar, and underwater detonations, for more than a generation. Although we do not know if more humpback whales might have used the action area or the reproductive success of humpback whales in the North Atlantic Ocean would be higher absent their exposure to these activities, the rate at which humpback whales occur in the Gulf of Maine suggests that humpback whale numbers have increased substantially in these important calving areas despite exposure to these stress regimes. Given this, we do not expect that BOEM and BSEE’s proposed seismic and HRG surveys are likely to affect the rate at which humpback whales, in the North Atlantic Ocean are increasing. Bowhead whales in the Arctic have been exposed to seismic surveys due to oil and gas exploration, oil and gas development and production and decommissioning activities since the late 1960s. Despite this exposure, the western Arctic stock of bowhead whales has been increasing at a rate of approximately 3.4 percent per year (George et al 2004).

Based on the evidence available, we conclude that the G&G activities BOEM and BSEE proposed in the mid-and South Atlantic Planning Areas from 2013-2020 are not likely to adversely affect the population dynamics, behavioral ecology, and social dynamics of individual humpback whales in ways or to a degree that would reduce their fitness. As we discussed in the *Approach to the Assessment* section of this Opinion, an action that is not likely to reduce the fitness of individual whales would not be likely to reduce the viability of the populations those individual whales represent (that is, we would not expect reductions in the reproduction, numbers, or distribution of those populations). As a result, we conclude that the G&G activities BOEM and BSEE propose from 2013 to 2020 would not appreciably reduce humpback whales' likelihood of surviving and recovering in the wild.

NORTH ATLANTIC RIGHT WHALES. We did not estimate the number of North Atlantic right whales that might be exposed to vessel traffic independent of the number of individuals that might be exposed to seismic and HRG surveys because the data we would have needed to support those analyses were not available.

Right whales would primarily be exposed to seismic and HRG survey activities in the mid-Atlantic Planning Area but could also be exposed to a degree during their seasonal migrations. Right whales may also be exposed to limited HRG surveys within their southeastern critical habitat off Georgia and Florida. Right whales use mid-Atlantic waters as a migratory pathway between the winter calving grounds and their spring and summer nursery and feeding areas in the Gulf of Maine. Right whales also use critical habitat in the southeast as nursing and calving habitat. Based on the BOEM's models, each year we would expect that numbers of North Atlantic right whales might be exposed to seismic and HRG surveys and be "taken" as a result of that exposure (see Tables 13 and 14 later in this section).

North Atlantic right whales are likely to respond to low frequency transmission from seismic surveys, however, the evidence is equivocal on whether North Atlantic right whales are likely to respond upon being exposed to the mid-frequency transmissions from HRG surveys, or the nature of any responses they might exhibit if they respond at all. The information available on right whales vocalizations suggests that right whales produce moans less than 400 Hz in frequency (Watkins and Schevill 1972; Thompson *et al.* 1979; Spero 1981). However, Nowacek *et al.* (2004) conducted controlled exposure experiments on North Atlantic right whales using ship noise, social sounds of conspecifics, and an alerting stimulus (frequency modulated tonal signals between 500 Hz and 4.5 kHz). Animals were tagged with acoustic sensors (D-tags) that simultaneously measured movement in three dimensions. Whales reacted strongly to alert signals at received levels of 133-148 dB SPL, mildly to conspecific signals, and not at all to ship sounds or actual vessels. Although the alert stimulus caused whales to immediately cease foraging behavior and swim rapidly to the surface, Nowacek *et al*. offer no information on whether the whales were probably responding to the low- or mid-frequency components of the signals.

Although North Atlantic right whales appear to be able to hear mid-frequency (1 kHz–10 kHz) sounds, the limited evidence available suggests that sounds in this frequency range appear to lie at the periphery of their hearing range. The tonal vocalizations right whales produce can be divided into simple, low-frequency, stereo-typed calls and more complex, frequency-modulated, higher-frequency calls (Parks and Clark 2007). Most of these sounds range in frequency from 0.02 to 15 kHz, with dominant frequency ranges from 0.02 to less than 2 kHz with some sounds having multiple harmonics (Parks and Tyack 2005). Assuming that right whales will focus their attentional resources on the frequency ranges of their vocalizations, right whales seem less likely to devote attentional resources to stimuli in the frequency ranges of mid-frequency sonar. As a result, they are not likely to respond physiologically or behaviorally to sounds in this frequency range.

Mother/calf pairs, and particularly mothers, are the mostly highly sensitive group to anthropogenic disturbance and respond more frequently than other sex, age, and activity classes. Calves are less experienced, possess fewer metabolic reserves, and are less capable of rapid travel than are their mothers, making responses such as rapid evasion or movement from the area a significant concern due to the potential for mother/calf separation and subsequent fitness consequences to the calf. Mothers are also sensitive to metabolic expenses, such as those involved with evasive behavior, due to the high demands placed upon them from simultaneous fasting and nursing. This means that exposure to stressors in critical habitat may produce greater impacts here to the subset of individuals most critical for the survival and recovery of the species than would be expected elsewhere.

North Atlantic right whales occurring within southeastern and mid-Atlantic U.S. SMAs and DMAs are likely to be migrating and possibly mating. These coastal corridors are established as areas to protect right whales along their migratory routes or in areas where temporarily high abundances occur. If migrating individuals were exposed to ephemeral sound sources like the ones proposed that elicited responses similar to those found in the *Response Analysis*, then we might expect a brief deflection from the direction of travel. Although this may entail a small metabolic expense and a possible stress response, we expect these would be so small as to not pose a concern for the individual’s overall fitness. We expect that individuals involved in mating or courtship would be highly motivated to continue this behavior and ignore a mild stimulus that would elicit a response in other behavioral states. Individuals are expected to attend to higher-level exposure, such as that which would be expected upon a vessel closely approaching with its acoustic equipment operating, and avoid potential collisions with a slow-moving vessel, such as those proposed for use.

Given the significance of right whale response, the reasonably likely exposure that right whales will receive while in critical habitat is critical to establish and minimize. BOEM provided detailed documentation as to the level of effort expected in and near critical habitat. This effort will be limited to a few surveys lasting a few days at most, although most would be one or two days. No more than two surveys using acoustic sources with frequencies within the audible range for right whales will be scheduled each year only if they are critical to the Renewable Energy or Marine Minerals programs. BOEM provided detailed acoustic characterization of these sources. This information led us to conclude that several sources would not be audible to right whales. Other sources could or would be audible, including vessel noise, boomer, sub-bottom profiler, and chirp seismic systems. BOEM also provided modeling results for acoustic sources that suggested that the potential for sound to propagate to 160 dB show less than ~100 m; some show < 10 meter (Martin et al 2012; Zykov and MacDonnell 2013).

However, use of these acoustic sources, and thus exposure to these sources, would occur very infrequently (up to twice annually). BOEM acknowledges that take could occur as a result of these activities in SMAs, DMAs and critical habitat collectively. A mother-calf pair, could be exposed during surveys within critical habitat but responses would be similar to those described in the *Response Analysis* (temporary behavioral responses consisting of movement away from the source). Based upon responses of mother/calf pairs to other acoustic stressors identified in the response analysis, we do not expect the source to be so disturbing as to result in extensive movements away such that energy demands on mothers or calves would produce a fitness consequence for the individuals.

Based on the evidence available, we conclude that the G&G activities BOEM and BSEE proposed in the mid-and South Atlantic Planning Areas from 2013-2020 are not likely to adversely affect the population dynamics, behavioral ecology, and social dynamics of individual North Atlantic right whales in ways or to a degree that would reduce their fitness. As we discussed in the *Approach to the Assessment* section of this Opinion, an action that is not likely to reduce the fitness of individual whales would not be likely to reduce the viability of the populations those individual whales represent (that is, we would not expect reductions in the reproduction, numbers, or distribution of those populations). As a result, we conclude that the G&G activities BOEM and BSEE proposed in the mid-and South Atlantic Planning Areas from

2013-2020 would not appreciably reduce the North Atlantic right whale's likelihood of surviving and recovering in the wild.

SEI WHALES. We did not estimate the number of sei whales that might be exposed to vessel traffic independent of the number of individuals that might be exposed to seismic and HRG surveys because the data we would have needed to support those analyses were not available.

Sei whales would primarily be exposed in spring and fall possibly in both Planning Areas to seismic and HRG surveys and primarily in the more offshore waters of the Action Area. In the western Atlantic Ocean, sei whales occur from Labrador, Nova Scotia, and Labrador in the summer months and migrate south to Florida and the northern Caribbean in the winter months. Based on the BOEM's models, each year we would expect that numbers of sei whales might be exposed to seismic and HRG surveys and be "taken" as a result of that exposure (see Tables 13 and 14 later in this section).

Like fin whales, sei whales in the Action Area seem likely to respond to the ship traffic associated with the activities the U.S. Navy plans to conduct in the Action Area in ways that approximate their responses to whale watch vessels. As discussed in the *Environmental Baseline* section of this Opinion, those responses are likely to depend on the distance of a whale from a vessel, vessel speed, vessel direction, vessel noise, and the number of vessels involved in a particular maneuver. Sei whales also seem most likely to try to avoid being exposed to the survey activities .We do not have the information necessary to determine which of the many sounds associated with an exercise is likely to trigger avoidance behavior in sei whales (for example, engine noise, airguns, electromechanical equipment or some combination of these) or whether sei whales would avoid being exposed to specific received levels, the entire sound field associated with a survey, or the general area in which a survey would occur.

Particular whales might not respond to the vessels, while in other circumstances, sei whales are likely to change their vocalizations, surface time, swimming speed, swimming angle or direction, respiration rates, dive times, feeding behavior, and social interactions (Amaral and Carlson 2005; Au and Green 2000, Cockeron 1995, Erbe 2002, Félix 2001, Magalhães *et al.* 2002, Richter *et al.* 2003, Scheidat *et al.* 2004, Simmonds 2005, Watkins 1986, Williams *et al.* 2002). Some of these whales might experience physiological stress (but not "distress") responses to seismic surveys and HRG surveys.

Based on the evidence available, we conclude that that the G&G activities BOEM and BSEE proposed in the mid-and South Atlantic Planning Areas from 2013-2020 are not likely to adversely affect the population dynamics, behavioral ecology, and social dynamics of individual sei whales in ways or to a degree that would reduce their fitness. As we discussed in the *Approach to the Assessment* section of this

Opinion, an action that is not likely to reduce the fitness of individual whales would not be likely to reduce the viability of the populations those individual whales represent (that is, we would not expect reductions in the reproduction, numbers, or distribution of those populations). As a result, we conclude that the G&G activities BOEM and BSEE proposed in the mid-and South Atlantic Planning Areas from 2013-2020 and the activities they would authorize each year would not appreciably reduce the sei whale's likelihood of surviving and recovering in the wild.

SPERM WHALES. Sperm whales would be exposed to seismic and HRG survey activity primarily in the fall, winter and spring in the mid-Atlantic Planning Area and would migrate to northern Areas outside of the Action Area in the spring/summer to feed.

If exposed to low- and mid-frequency seismic and HRG transmissions, sperm whales are likely to hear and respond to those transmissions. The only data on the hearing range of sperm whales are evoked potentials from a stranded neonate (Carder and Ridgway 1990). These data suggest that neonatal sperm whales respond to sounds from 2.5-60 kHz. Sperm whales also produce loud broad-band clicks from about 0.1 to 20 kHz (Weilgart and Whitehead 1993, 1997; Goold and Jones 1995). These have source levels estimated at 171 dB re 1 Pa (Levenson 1974). Current evidence suggests that the disproportionately large head of the sperm whale is an adaptation to produce these vocalizations (Norris and Harvey 1972; Cranford 1992; but see Clarke 1979). This suggests that the production of these loud low frequency clicks is extremely important to the survival of individual sperm whales. The function of these vocalizations is relatively well-studied (Weilgart and Whitehead 1993, 1997; Goold and Jones 1995). Long series of monotonous regularly spaced clicks are associated with feeding and are thought to be produced for echolocation. Distinctive, short, patterned series of clicks, called codas, are associated with social behavior and interactions within social groups (Weilgart and Whitehead 1993).

Based on the frequencies of their vocalizations, which overlap the frequency range of mid-frequency acoustic sources, HRG transmissions might temporarily reduce the active space of sperm whale vocalizations. Airguns also produce frequencies up to 2,000 Hz, however, these frequencies are not the frequencies with most energy and should not reduce the active space of a sperm whale unless the whale is in close proximity to the source, if at all. Most of the energy of sperm whales clicks is concentrated at 2 to 4 kHz and 10 to 16 kHz, which overlaps with the mid-frequency transmission of HRG equipment. Other studies indicate sperm whales' wide-band clicks contain energy between 0.1 and 20 kHz (Weilgart and Whitehead 1993, Goold and Jones 1995). Ridgway and Carder (2001) measured low-frequency, high amplitude clicks with peak frequencies at 500 Hz to 3 kHz from a neonate sperm whale.

There is some evidence of disruptions of clicking and behavior from echosounders and submarine sonars (Goold 1999, Watkins and Scheville1975, Watkins *et al.* 1985), pingers (Watkins and Scheville 1975), the Heard Island Feasability Test (Bowles *et al.* 1994), and the Acoustic Thermometry of Ocean Climate (Costa *et al.*1998). Sperm whales have been observed to frequently stop echolocating in the presence of underwater pulses made by echosounders (Watkins and Scheville 1975). Goold (1999) reported six sperm whales that were driven through a narrow channel using ship noise, echosounder, and fishfinder emissions from a flotilla of 10 vessels. Watkins and Scheville (1975) showed that sperm whales interrupted click production in response to pinger (6 to 13 kHz) sounds. They also stopped vocalizing for brief periods when codas were being produced by other individuals, perhaps because they can hear better when not vocalizing themselves (Goold and Jones 1995).

As discussed previously, sperm whales have been reported to have reacted to military sonar, apparently produced by a submarine, by dispersing from social aggregations, moving away from the sound source, remaining relatively silent and becoming difficult to approach (Watkins *et al.* 1985). Captive bottlenose dolphins and a beluga whale exhibited changes in behavior when exposed to 1 sec pulsed sounds at frequencies similar to those emitted by multi- beam sonar that is used in geophysical surveys (Ridgway *et al.* 1997, Schlundt *et al.* 2000), and to shorter broadband pulsed signals (Finneran *et al.* 2000, 2002). Behavioral changes typically involved what appeared to be deliberate attempts to avoid the sound exposure or to avoid the location of the exposure site during subsequent tests (Schlundt *et al.* 2000, Finneran *et al.* 2002). Dolphins exposed to 1-sec intense tones exhibited short-term changes in behavior above received sound levels of 178 to 193 dB re 1 Pa rms and belugas did so at received levels of 180 to 196 dB and above. Received levels necessary to elicit such reactions to shorter pulses were higher (Finneran *et al.* 2000, 2002). Test animals sometimes vocalized after exposure to pulsed, mid-frequency sound from a watergun (Finneran *et al.* 2002). In some instances, animals exhibited aggressive behavior toward the test apparatus (Ridgway *et al.* 1997, Schlundt *et al.* 2000). The relevance of these data to free-ranging odontocetes is uncertain. In the wild, cetaceans sometimes avoid sound sources well before they are exposed to the levels listed above, and reactions in the wild may be more subtle than those described by Ridgway *et al.* (1997) and Schlundt *et al.* (2000).

Published reports identify instances in which sperm whales may have responded to an acoustic source and other instances in which they did not appear to respond behaviorally when exposed to seismic surveys. Mate *et al.* (1994) reported an opportunistic observation of the number of sperm whales to have decreased in an area after the start of airgun seismic testing. However, Davis *et al.* (2000) noted that sighting frequency did not differ significantly among the different acoustic levels examined in the northern Gulf of Mexico, contrary to what Mate *et al.* (1994) reported. In one DTAG deployment in the northern Gulf of

Mexico on July 28, 2001, researchers documented that the tagged whale moved away from an operating seismic vessel once the seismic pulses were received at the tag at roughly 137 dB re 1 Pa (Johnson and Miller 2002). Sperm whales may also have responded to seismic airgun sounds by ceasing to call during some (but not all) times when seismic pulses were received from an airgun array >300 km away (Bowles *et al.* 1994).

A recent study offshore of northern Norway indicated that sperm whales continued to call when exposed to pulses from a distant seismic vessel. Received levels of the seismic pulses were up to 146 dB re 1 Pa peak-to-peak (Madsen *et al.* 2002). Similarly, a study conducted off Nova Scotia that analyzed recordings of sperm whale sounds at various distances from an active seismic program did not detect any obvious changes in the distribution or behavior of sperm whales (McCall Howard 1999). Recent data from vessel-based monitoring programs in United Kingdom waters suggest that sperm whales in that area may have exhibited some changes in behavior in the presence of operating seismic vessels (Stone 1997, 1998, 2000, 2001, 2003). However, the compilation and analysis of the data led the author to conclude that seismic surveys did not result in observable effects to sperm whales (Stone 2003). The results from these waters seem to show that some sperm whales tolerate seismic surveys.

Preliminary data from an experimental study of sperm whale reactions to seismic surveys in the Gulf of Mexico and a study of the movements of sperm whales with satellite-linked tags in relation to seismic surveys show that during two controlled exposure experiments in which sperm whales were exposed to seismic pulses at received levels up to 148 dB re 1 Pa over octave band with most energy, the whales did not avoid the vessel or change their feeding efficiency (National Science Foundation 2003). In a later publication concerning the experiments, Madsen *et al.* (2006) report that seven of eight tagged sperm whales continued to perform foraging dives throughout exposure to seismic airguns at levels up to 147 dB (rms) (the eighth whale remained at the surface during exposure). Although the sample size is small (4 whales in 2 experiments), the results are consistent with those off northern Norway (Madsen *et al.* 2002).

Other studies identify instances in which sperm whales did not respond to anthropogenic sounds. Sperm whales did not alter their vocal activity when exposed to levels of 173 dB re 1 Pa from impulsive sounds produced by 1 g TNT detonators (Madsen and Mohl 2000). Richardson *et al.* (1995) citing a personal communication with J. Gordon suggested that sperm whales in the Mediterranean Sea continued calling when exposed to frequent and strong military sonar signals. When Andre *et al.* (1997) exposed sperm whales to a variety of sounds to determine what sounds may be used to scare whales out of the path of vessels, sperm whales were observed to have startle reactions to 10 kHz pulses (180 db re 1 Pa at the source), but not to the other sources played to them.

These studies suggest that the behavioral responses of sperm whales to anthropogenic sounds are highly variable, but do not appear to result in the death or injury of individual whales or result in reductions in the fitness of individuals involved. Responses of sperm whales to anthropogenic sounds probably depend on the age and sex of animals being exposed, as well as other factors. There is evidence that many individuals respond to certain sound sources, provided the received level is high enough to evoke a response, while other individuals do not.

The evidence available suggests that sperm whales are likely to detect low and mid-frequency seismic and HRG transmissions. In most circumstances, sperm whales are likely to try to avoid that exposure or are likely to avoid specific areas. For example, sperm whales have moved out of areas after the start of air gun seismic testing (Davis *et al.* 1995). Those sperm whales that do not avoid the sound field might interrupt communications, echolocation, or foraging behavior. In either case, sperm whales that avoid these sound fields, stop communicating, echolocating or foraging might experience significant disruptions of normal behavior patterns that are essential to their individual fitness. We do not, however, expect these disruptions to result in the death or injury of any individual animal or to result in physiological stress responses that rise to the level of distress.

Like fin and sei whales, individual sperm whales are also likely to respond to the ship traffic associated with the maneuvers might approximate their responses to whale watch vessels. As discussed in the *Environmental Baseline* section of these Opinions, those responses are likely to depend on the distance of a whale from a vessel, vessel speed, vessel direction, vessel noise, and the number of vessels involved in a particular maneuver. The closer sperm whales are to these maneuvers and the greater the number of times they are exposed, the greater their likelihood of being exposed and responding to that exposure. Particular whales might not respond to the vessels, while in other circumstances, sperm whales are likely to change their vocalizations, surface time, swimming speed, swimming angle or direction, respiration rates, dive times, feeding behavior, and social interactions (Amaral and Carlson 2005; Au and Green 2000, Cockeron 1995, Erbe 2002, Félix 2001, Magalhães *et al.* 2002, Richter *et al.* 2003, Scheidat *et al.* 2004, Simmonds 2005, Watkins 1986, Williams *et al.* 2002). Some of these whales might experience physiological stress (but not “distress”) responses. However, we do not expect these responses to continue long enough to have fitness consequences for individual sperm whales because these whales are likely to have energy reserves sufficient to meet the demands of their normal behavioral patterns and those of a stress physiology.

Based on the evidence available, we conclude that the G&G activities BOEM and BSEE propose to authorize in the mid-and South Atlantic Planning Areas from 2013-2020 are not likely to adversely affect the population dynamics, behavioral ecology, and social dynamics of individual sperm whales in ways or to a degree that would reduce their fitness. As we discussed in the *Approach to the Assessment* section of this Opinion, an action that is not likely to reduce the fitness of individual sperm whales is not likely to reduce the viability of the populations those individual whales represent by reducing the population dynamics, behavioral ecology, and social dynamics of those populations (that is, we would not expect reductions in the reproduction, numbers, or distribution of those populations). As a result, we conclude that the G&G activities BOEM and BSEE proposed in the mid-and South Atlantic Planning Areas from 2013-2020 and the G&G activities BOEM and BSEE conducts each year would not appreciably reduce the sperm whale's likelihood of surviving and recovering in the wild.

Table 13: Expected number of instances in which individual members of endangered whale species are likely to be "taken" as a result of their exposure to seismic airgun and HRG survey activities. **See the Approach to the Assessment and the "Exposure Analyses" and "Response Analyses" subsections of the Effects of the Action section of this Opinion for an explanation of the methods used to produce these estimates.**

Species	Annual Take Estimates from Surveys Using Airguns Under the Oil and Gas Program (2013 – 2020)									8-Year Estimate Based on Mean Annual Estimate and Confidence Intervals based on t-distribution
	2012	2013	2014	2015	2016	2017	2018	2019	2020	Total Takes Over 8 Years (95% Confidence Interval)
Harass										
North Atlantic Right Whale	0	114	224	26	185	137	127	86	58	851 (379-1,323)
Blue Whale	0	214	419	44	270	196	151	86	53	1,274 (413-2,135
Fin Whale	0	431	847	93	578	426	341	189	120	2,668 (937-4,439)
Sei Whale	0	193	378	41	251	184	145	80	49	1,175 (394-1,955)
Humpback Whale	0	578	1,131	118	719	521	396	227	136	3,402 (1,077-5,726)
Sperm Whale	0	15,567	30,356	2,980	17,549	12,443	8,756	5,364	2,926	85,281 (23,115-147-447)

Table 14: Expected number of instances in which individual members of endangered whale species are likely to be "taken" as a result of their exposure to Non-airgun HRG survey activities under the Renewable Energy and Marine Minerals Programs. See the Approach to the Assessment and the "Exposure Analyses" and "Response Analyses" subsections of the Effects of the Action section of this Opinion for an explanation of the methods used to produce these estimates.

Species	Annual Take Estimates Using Non-Airgun HRG Surveys (2013 - 2020)									8-Year Estimate Based on Mean Annual Estimate and Confidence Intervals
	2012	2013	2014	2015	2016	2017	2018	2019	2020	Total Takes Over 8 Years (Confidence Interval)
Harass										
North Atlantic Right Whale	0	0	0	0	0	0	1	0	1	3 (2-5)
Blue Whale	0	0	0	0	0	0	0	0	0	1 (1-2)
Fin Whale	0	0	0	0	0	0	0	0	1	2 (1-3)
Sei Whale	0	0	0	0	0	0	0	0	0	(1-2)
Humpback Whale	0	0	0	0	0	0	0	0	1	3 (3-4)
Sperm Whale	0	0	0	0	0	0	4	4	12	19 (0-45)

SEA TURTLES.

The information available has not allowed us to estimate the probability of the different sea turtles being exposed to vessel traffic, seismic or HRG survey activities associated with the G&G activities BOEM and BSEE proposed in the mid-and South Atlantic Planning Areas from 2013-2020.

Further, the information on the hearing capabilities of sea turtles is limited, although the information available suggests that the auditory capabilities of sea turtles are centered in the low-frequency range (<1 kHz) (Ridgway *et al.* 1969; Lenhardt *et al.* 1983; Bartol *et al.* 1999, Lenhardt 1994, O'Hara and Wilcox 1990). Ridgway *et al.* (1969) studied the auditory evoked potentials of three green sea turtles (in air and through mechanical stimulation of the ear) and concluded that their maximum sensitivity occurred from 300 to 400 Hz with rapid declines for tones at lower and higher frequencies. They reported an upper limit for cochlear potentials without injury of 2000 Hz and a practical limit of about 1000 Hz. This is similar to estimates for loggerhead sea turtles, which had most sensitive hearing between 250 and 1000 Hz, with rapid decline above 1000 Hz (Bartol *et al.* 1999). No audiometric data are available for leatherback sea turtles. We assume, however, that all sea turtle hearing sensitivities will be similar to those of green and loggerhead sea turtles with their best hearing sensitivity in the low frequency range: from 50 to 400 Hz with rapid declines for tones at lower and higher frequencies and a practical upper limit of about 1600 Hz in water (Piniak *et al*. 2012).

Because the side scan sonar, depth sounders and subbottom profilers (at frequencies above 2 kHz) that would be used during the proposed G&G activities transmit at frequencies above hearing thresholds for sea turtles, sea turtles that are exposed to those transmissions are not likely to respond to that exposure. As a result, HRG surveys using those sound sources ―may affect, but are not likely to adversely affect green, hawksbill, Kemp's ridley, leatherback, or Northwest Atlantic loggerhead sea turtles.

Available studies suggest some sea turtles exhibit an avoidance reaction to airgun-generated sounds. McCauley *et al.* (2000b; 2000a) investigated the effects of airguns on sea turtle behavior. The authors found that green and loggerhead sea turtles show avoidance to airgun arrays at 2 km (1.1 nm) and at 1 km (0.54 nm) with received levels of 166 dB re 1 µPa and 175 dB re 1 µPa, respectively. Individual sea turtles responded consistently by noticeably increasing swimming activity above a level of approximately 166 dB re 1 µPa (rms), as compared to swimming during non-airgun operation periods. The increase in swimming behavior tracked the received airgun level, by increasing at increasing levels. Above 175 dB re 1 µPa, turtle behavior became more erratic, possibly indicating the turtles were in an agitated state. In studies by Lenhardt (1994) and Lenhardt *et al.* (1983), loggerhead and Kemp's ridley turtles responded to airgun pulses and low-frequency sound – e.g., by becoming active and swimming to the surface upon exposure. Moein *et al*. (1994) used an evoked potential method to test sea turtle hearing and reported avoidance behavior in loggerhead sea turtles at the beginning of airgun exposure trials; however, repeated airgun exposures days after the initial tests did not elicit a statistically significant avoidance response. The authors concluded this may be due to either habituation or temporary threshold shift in the turtles hearing capability. Based on a review of sea turtle data from 11 L-DEO seismic surveys since 2003, Holst *et al*. (2006) concluded that turtles exhibited localized avoidance during both large- and small-source seismic surveys.

Although studies suggest sea turtles are most likely to avoid seismic airgun pulses, monitoring reports from seismic surveys indicate occasions when sea turtles were likely exposed to seismic airgun pulses – e.g., green, leatherback, and olive ridley turtles during surveys in the Hess Deep area of the eastern tropical Pacific Ocean (Smultea and Holst 2003 as cited in NMFS, 2006h). Of six sea turtles that were sighted during those surveys, five were seen while airguns were active. In at least one instance, an olive ridley sea turtle was sighted within ten meters of the array while active. This turtle was reported to exhibit visible responses to either exposure to the seismic pulses or to the physical presence of the array and floats. Although the turtle swam away from the vessel and was not reported to suffer physical injury, it is assumed the turtle experienced a stress response to its exposure that may have risen to the level of harassment. Similarly, Holst *et al.* (2005b) report during seismic operations in the southern Gulf of Mexico off the Yucatán Peninsula, that seven sea turtles were sighted within the 180 dB safety radius in shallow water (< 40 m). One of these turtles was reported to be actively swimming away from the seismic source. Six of these turtles were seen < 200 m from the operating airguns before the airguns were powered- or shut-down (Holst, et al., 2005b).

A more recent study by DeRuiter and Doukara (2012) reported on the observed reactions of loggerhead sea turtles during a seismic survey in the Mediterranean Sea off Algeria. The seismic survey was conducted with a 13-airgun array (nominal source level of 252 dB re 1µPa at 1 m (peak)), towed at a depth of 11.5m at 5 kts. All but 3 (swimming underwater) of the 164 loggerheads observed (98 percent), were motionless at the surface when observed and ranged from 10s of meters to nearly 2 km from the vessel when sighted. Observed reactions included head raising (6), flipper agitation (7), air bubble production around head (2) or diving. Of the 86 turtles whose dive behavior could be visually tracked until their passage more than 100 m behind the array, 49 (57 percent) dove (dove before passing behind the airgun array) and 37 (43 percent) did not dive (were in view at the surface until they had passed more than 100 m behind the airgun array). At least 6 turtles dove immediately following an airgun shot indicating a startle response. The authors suggest that loggerhead turtles interrupted basking behavior and dove (an avoidance response) in response to airgun sounds at relatively close ranges (130 m median distance, 839 m maximum distance) and exposure levels estimated at about 191 dB re 1µPa at 1 m (peak) at 130 m and 175 dB re 1µPa at 1 m (peak) at 839 m.

Sea turtle hearing thresholds appear to be higher than those for mammals and DFO (2004) concluded it is unlikely that sea turtles would be more sensitive to seismic operations than cetaceans, based on available studies. Sea turtles are expected to be less sensitive to sounds; however, behavioral responses to environmental sounds are documented in several controlled experiments. Some possible reactions to low frequency sounds include startle responses and rapid swimming (M. Lenhardt, 2002; McCauley 2001 as cited in NMFS, 2006h), as well as swimming towards the surface at the onset of the sound (M. L. Lenhardt, 1994).

Seismic airgun surveys conducted off of heavily used nesting beaches during the nesting season could temporarily displace adult turtles that are approaching or departing nesting beaches or resting in offshore waters between nesting events. Beaches of southeast Florida have been identified as the most important nesting area for loggerhead turtles The northern segment of the Archie Carr NWR borders the Action Area, and it has been estimated that 25 percent of all loggerhead nesting in the U.S. occurs there (USDOI, FWS, 2011). It is likely that large numbers of sea turtles would be present in nearshore and inner shelf during the nesting season. Breeding adults and nesting adult females could be exposed to airgun seismic survey-related sounds. Hatchlings may be somewhat insulated from the highest sound levels because of their occurrence at or near the sea surface. However, avoidance behavior may shorten the exposure period, and the ramping-up of airguns during the proposed activities would provide opportunity for avoidance by sea turtles, thereby minimizing exposure to received levels of concern. Avoidance or any disruptions in sea turtle behavior are expected to be temporary and are not expected to cause any injury. For those turtles that might be exposed to seismic pulses at levels above 166 dB re 1 µPa during the proposed activities, we expect this could result in a stress response.

Nevertheless, we conclude that the G&G activities BOEM and BSEE proposed in the mid-and South Atlantic Planning Areas from 2013-2020 are not likely to reduce the viability of the nesting aggregations those sea turtles represent by reducing the population dynamics, behavioral ecology, and social dynamics of those populations (that is, we would not expect reductions in the reproduction, numbers, or distribution of those populations). As a result, those activities would not be expected to appreciably reduce the likelihood of green, hawksbill, Kemp’s ridley, leatherback, or Northwest Atlantic loggerhead sea turtles surviving and recovering in the wild by reducing their reproduction, numbers, or distribution.

Atlantic Sturgeon

The information available has not allowed us to estimate the probability of Atlantic sturgeon being exposed to vessel traffic, seismic or HRG survey activities associated with the G&G activities BOEM and BSEE proposed in the mid-and South Atlantic Planning Areas from 2013-2020. Because of their coastal distribution, only subadult and adult Atlantic sturgeon would be exposed to HRG surveys. Exposure to seismic surveys can occur since they can be conducted within 50 m of water; however, it is unclear how much ambient sound levels will increase in the shallow, coastal environment.

Sturgeon in general can be classified as fishes that detect sounds from below 50 Hz to perhaps 800-1,000 Hz (though several probably only detect sounds to 600-800 Hz). These fishes have a swim bladder but no known structures in the auditory system that would enhance hearing, and sensitivity (lowest sound detectable at any frequency) is not very great. Sounds would have to be more intense to be detected compared to fishes with swim bladders that enhance hearing. Sturgeon can detect both particle motion and pressure.

Vessel noise produces sounds in the general hearing range of fishes (Amoser et al., 2004). Additionally, vessels (i.e., trawlers, ferries, small boats) can change fish behavior (e.g., induce avoidance, alter swimming speed and direction, and alter schooling behavior) (Sarà et al., 2007). The sounds produced by motor-driven ships causes herring to dive and swim away from the vessel (Mitson and Knudsen, 2003). Sand et al. (2008) have pointed out that passing ships produce high levels of infrasonic and low-frequency noise (>10-1,000 Hz), and that infrasonic frequencies may be responsible for the observed avoidance reactions.

Atlantic sturgeon are expected to avoid seismic or HRG sources, but some may not avoid the source vessel and may be exposed to seismic sound at levels of concern. Avoidance behavior may shorten the exposure period. Avoidance or any disruptions in Atlantic sturgeon behavior are expected to be temporary and are not expected to cause any injury. Stress responses that rise to the level of harassment are not expected.

Conclusion

After reviewing the current status of blue, fin, sei, humpback, north Atlantic right and sperm whales, green, hawksbill, Kemp's ridley, leatherback and Northwest Atlantic loggerhead sea turtles and Atlantic sturgeon; the environmental baseline for the Action Area; the anticipated effects of the proposed activities; and the cumulative effects, it is NMFS' biological opinion that the proposed action, as described in this Opinion, is not likely to jeopardize the continued existence of these species. It is also NMFS' opinion that the proposed action, as described is not likely to destroy or adversely modify designated North Atlantic right whale critical habitat.

INCIDENTAL TAKE STATEMENT

Section 9 of the ESA and federal regulation pursuant to section 4(d) of the ESA prohibit the “take” of endangered and threatened species, respectively, without special exemption. “Take” is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct. Harm is further defined by NMFS to include significant habitat modification or degradation that results in death or injury to listed species by significantly impairing essential behavioral patterns, including breeding, feeding, or sheltering. Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. 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 (ITS).

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

Section 7(b)(4)(C) of the ESA provides that if an endangered or threatened marine mammal is involved, the taking must first be authorized by Section 101(a)(5) of the Marine Mammal Protection Act of 1972, as amended (MMPA). Accordingly, this opinion does not exempt incidental take for North Atlantic right whales (*Eubalaena glacialis*), blue whales (*Balaenoptera musculus*), fin whales (*Balaenoptera physalus*), humpback whales (*Megaptera novaeangliae*) sei whales (*Balaenoptera borealis*) and sperm whales (*Physeter macrocephalus*), at this time. In order to be exempt from the Section 9 take prohibitions, an MMPA authorization to take those marine mammals is required. NMFS will reinitiate this consultation as appropriate on NMFS Permits and Conservation Division’s proposed issuance of any MMPA incidental take authorization and may exempt the take of listed marine mammals at that time.

Based on the analyses contained in the Effects of the Action section of this Opinion, NMFS anticipates the incidental harassment of blue whales (*Balaenoptera musculus*), fin whales (*Balaenoptera physalus*), humpback whales (*Megaptera novaeangliae*), North Atlantic right whales (*Eubalaena glacialis),* sei whales (*Balaenoptera borealis*) and sperm whales (*Physeter macrocephalus*), as well as green sea turtles (*Chelonia mydas*), hawksbill sea turtles (*Eretmochelys imbricata*), Kemp's ridley sea turtles (*Lepidochelys kempii*), leatherback sea turtles (*Dermochelys coriacea*), and Northwest Atlantic loggerhead sea turtles (*Caretta caretta*) as a result of the proposed seismic and HRG activities.

Amount or Extent of Take

The 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 the proposed action 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).

The effects analysis contained in this Opinion concluded that individual blue whales, fin whales, humpback whales, North Atlantic right whales, sei whales and sperm whales; green sea turtles, hawksbill sea turtles, Kemp's ridley sea turtles, leatherback sea turtles and Northwest Atlantic loggerhead sea turtles have small probabilities of being exposed to sound fields associated with seismic and HRG equipment or noise and other environmental cues associated with the movement of survey and support vessels. When we concluded that an exposure was likely to result in evasive behavior or changes in behavioral state, we treated those behavioral responses as "harassment" for the purposes of this incidental take statement.

Based on our analyses, we did not expect any whales, sea turtles or sturgeon to be struck by survey vessels or entangled in survey equipment, so this Incidental Take Statement does not exempt "take" resulting from vessel strikes or entanglement in equipment.

The instances of harassment identified in Table 1 and Table 2 are expected to result from changes from foraging, resting, milling, and other behavioral states that require lower energy expenditures to traveling, avoidance, and behavioral states that require higher energy expenditures and, therefore, would represent significant disruptions of the normal behavioral patterns of the animals that have been exposed. No whales are likely to die or be wounded as a result of their exposure to the G&G activities BOEM and BSEE propose from 2013 to 2020. Therefore, for the purposes of this biological opinion and incidental take statement, we assume that the G&G activities BOEM and BSEE propose to conduct within and adjacent to the mid-Atlantic and South Atlantic Planning Areas are likely to result in the following incidental "take" (by harassment) due to seismic and HRG survey activities under the Oil and Gas program (Table 1) and under the Renewable Energy and Marine Minerals Programs (Table 2):

Table 1: Expected number of instances in which individual members of endangered whale species are likely to be "taken" (by harassment) as a result of their exposure to seismic survey activities under the Oil and Gas Program. See the Approach to the Assessment and the "Exposure Analyses" and "Response Analyses" subsections of the Effects of the Action section of this Opinion for an explanation of the methods used to produce these estimates.

Species	Annual Take Estimates from Seismic and HRG Surveys Using Airguns the Oil and Gas Program (2013 – 2020)								8-Year Estimate (Mean Annual Estimate based on t-distribution) (Confidence Interval)
	2013	2014	2015	2016	2017	2018	2019	2020	Total Takes Over the 8-Year Duration of the Proposed Action (95% Confidence Interval)
Harass									
North Atlantic Right Whale	114	224	26	185	137	127	86	58	851 (379-1,323)
Blue Whale	214	419	44	270	196	151	86	53	1,274 (413-2,135)
Fin Whale	431	847	93	578	426	341	189	120	2,688 (937-4,439)
Sei Whale	193	378	41	251	184	145	80	49	1,175 (394-1,955)
Humpback Whale	578	1,131	118	719	521	396	227	136	3,402 (1,077-5,726)
Sperm Whale	15,567	30,356	2,980	17,549	12,443	8,756	5,364	2,926	85,281 (23,115-147,447)

Table 2 : Expected number of instances in which individual members of endangered whale species are likely to be "taken" (by harassment) as a result of their exposure to Non-Airgun HRG survey activities under the Renewable Energy and Marine Minerals Programs. See the Approach to the Assessment and the "Exposure Analyses" and "Response Analyses" subsections of the Effects of the Action chapter of this Opinion for an explanation of the methods used to produce these estimates.

Species	Annual Take Estimates Using Non-Airgun HRG Surveys for the Renewable and Marine Minerals Programs (2013 - 2020)								8-Year Estimate (Mean Annual Estimate based on t-distribution) (Confidence Interval)
	2013	2014	2015	2016	2017	2018	2019	2020	Total Takes Over the 8-Year Duration of the Proposed Action (95% Confidence Interval)

									Harass
North Atlantic Right Whale	**0**	**0**	**0**	**0**	**0**	**1**	**0**	**1**	**3** **(2-5)**
Blue Whale	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**1** **(1-2)**
Fin Whale	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**1**	**2** **(1-3)**
Sei Whale	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**1** **(1-2)**
Humpback Whale	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**1**	**3** **(3-4)**
Sperm Whale	**0**	**0**	**0**	**0**	**0**	**4**	**4**	**12**	**19** **(0-45)**

Determining the take of marine mammals requires knowledge of density of each endangered whale species in the areas where seismic and HRG surveys would occur, propagation of the sound fields in space and time, survey effort in tracklines or hours and spatial distribution of surveys. Because density estimates of blue whales, fin whales, humpback whales, North Atlantic right whales, sei whales and sperm whales in the survey area are uncertain; "take" of these species will have been exceeded if the density estimates used to produce these "take" estimates are exceeded, if the monitoring program associated with seismic and HRG survey activities detects any individuals of these species that have been harmed, wounded, or killed as a result of exposure to seismic and HRG transmissions, if survey effort (in number of tracklines or hours) is exceeded or if the spatial distribution of surveys change.

We also expect the proposed action might also take (by harassment) individual green, hawksbill, Kemp's ridley, leatherback and Northwest Atlantic loggerhead sea turtles as a result of exposure to acoustic energy during seismic and HRG surveying using boomer and sparkers. Harassment of green, hawksbill, Kemp's ridley, leatherback and Northwest Atlantic loggerhead sea turtles is expected to occur at received levels of seismic energy above 166 dB re 1 µPa. Available studies suggest some sea turtles show avoidance to airgun arrays with received levels of 166 dB re 1 µPa. This incidental take would result from exposure to acoustic energy during seismic surveys and HRG surveys using boomers and sparkers, would be in the form of harassment, and is not expected to result in the death or injury of any individuals that are exposed.

Determining the take of sea turtles requires knowledge of density of each sea turtle species in the areas where seismic surveys would occur, propagation of the sound field in space and time, survey effort in tracklines or hours and spatial distribution of surveys. Because density estimates of green, hawksbill, Kemp’s ridley, leatherback and Northwest Atlantic loggerhead sea turtles in the survey area are unknown; we cannot reliably quantify the anticipated amount of take of sea turtles during seismic surveys. “Take” of these species will have been exceeded if the monitoring program associated with seismic and HRG survey activities detects any individuals of these species that have been harmed, injured, or killed as a result of exposure to seismic and HRG transmissions, if survey effort (in number of tracklines or hours) are exceeded or if the spatial distribution of surveys change.

While Atlantic sturgeon may be disturbed by survey activities, we do not expect that this disturbance will rise to the level of harassment. We do not expect green, hawksbill, Kemp’s ridley, leatherback or northwest Atlantic loggerhead sea turtles to be taken by operation of side-scan sonars, chirp, subbottom-profilers, and single beam, swath or multibeam depth sounders as these are outside of the hearing ranges of these sea turtles. However, if overt adverse reactions (for example, dive reactions, or rapid departures from the area) by listed sea turtles are observed while these HRG sources are in operation, this may constitute take that is not covered in this Incidental Take Statement. If such overt adverse reactions are observed BOEM and/or BSEE must contact the Chief, Endangered Species Act Interagency Cooperation Division within 48 hours of the incident at 301-427-8403 and/or by email to kellie.foster-taylor@noaa.gov to determine whether reinitiation of consultation is required.

Any incidental “take” of blue, fin, humpback, North Atlantic right, sei or sperm whales, or green, hawksbill, Kemp’s ridley, leatherback and Northwest Atlantic loggerhead sea turtles is restricted to the action as proposed. If the incidental “take” meets or exceeds the predicted level as specified in this ITS by increases in survey effort in tracklines or hours or changes in the spatial distribution of surveys where surveys are not evenly distributed throughout the Action Area as BOEM has assumed in their modeling, BOEM and BSEE must reinitiate consultation.

Effect of the Take

In the accompanying Opinion, NMFS determined that this level of harassment is not likely to jeopardize the continued existence of the endangered or threatened species for which “take” (by harassment) would be exempted by this Incidental Take Statement. Studies of marine mammals, sea turtles and seismic and HRG equipment transmissions have shown behavioral responses by these species to airgun and sonar transmissions. Although the biological significance of the animal’s behavioral responses remains uncertain, the best scientific and commercial data available leads us to conclude that exposing these endangered and threatened species to seismic and HRG equipment transmissions might disrupt one or more behavioral patterns that are essential to an individual animal’s life history or to the animal’s contribution to a population. For the proposed action, behavioral responses that result from seismic and HRG equipment transmissions and any associated disruptions are expected to be temporary and would not affect the reproduction, survival, or recovery of these species.

Reasonable and Prudent Measures

NMFS believes the reasonable and prudent measures described below are necessary and appropriate to minimize the impacts of incidental take on threatened and endangered species:

1. BOEM shall condition the permits or leases it issues as part of the proposed action to require that each seismic or HRG vessel operator obtain an authorization under section 101(a)(5) of the MMPA as necessary and that this authorization be provided by the operator to BOEM prior to commencing activities under a BOEM permit or authorization.

2. BOEM and/or BSEE shall condition the permits or leases it issues as part of the proposed action to require operators to implement the mitigation measures described in the Terms and Conditions of this Opinion, as appropriate.

3. BOEM and BSEE must monitor the effectiveness of the mitigation measures described in the Terms and Conditions of this ITS. As methods to standardize data collection methods are available, BOEM will insure that data its collects or requires operators to collect comply with those standards (see NOAA Fisheries Service 2012 National Standards for a Protected Species Observer and Data Management Program: A model for Seismic Surveys (Baker et al, unpublished)) across survey types for moving (i.e., 2D, 3D), stationary and continuous sound sources (i.e., VSP surveys, shallow test and COST well drilling) and Planning Areas to allow for comparison between differences in survey design and different geographic locations.

4. BOEM and/or BSEE shall submit reports that identify the location, timing, number of hours, and other aspects of the seismic and HRG activities conducted during seismic and HRG survey activities that describe the implementation and results of monitoring and mitigation. These reports will also be standardized as methods become available (see NOAA Fisheries Service 2012 National Standards for a Protected Species Observer and Data Management Program: A model for Seismic Surveys (Baker et al, unpublished).

Terms and Conditions

To be exempt from the prohibitions of section 9 of the ESA, BOEM and BSEE must comply with the following terms and conditions, which implement the Reasonable and Prudent Measures described above and outlines the mitigation, monitoring and reporting measures required by the section 7 regulations (50 CFR 402.14(i)). These terms and conditions are non-discretionary. If BOEM and/or BSEE 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.

1. BOEM shall condition the permits or leases it issues as part of the proposed action to require each seismic or HRG vessel operator to obtain an authorization under section 101(a)(5) of the MMPA as necessary and that this authorization be provided by the operator to BOEM prior to commencing activities under a BOEM permit or authorization.

General Mitigation and Monitoring Requirements

1. Airgun Surveys: BOEM and BSEE shall not authorize any surveys using airguns within North Atlantic right whale critical habitat in the southeast United States from November 15 through April 15 of each year or within the Southeast and Mid-Atlantic seasonal management areas (SMAs) from November 1 through April 30 of each year or within any established Dynamic Management Areas (DMAs). Any surveys authorized outside of critical habitat, SMA boundaries or DMAs are required to remain at a distance such that received levels at these boundaries are no more than 160 dB re 1 μPa-m (rms) as determined by field verification or modeling.
2. All Surveys: All G&G survey vessel operators must comply with requirements for vessel strike avoidance. See Vessel Strike Avoidance below.
3. All Surveys: BOEM shall condition all G&G authorizations for shipboard surveys to include guidance for marine debris awareness which includes the safe disposal of trash and debris. All vessel operators, employees and contractors actively engaged in G&G surveys must be briefed on marine trash and debris awareness elimination as described in BSEE NTL No. 2012-G01 (*Marine Trash and Debris Awareness and Elimination*) (USDOI, BSEE, 2012a). G&G operators will be required to ensure that its employees and contractors are made aware of the environmental and socioeconomic impacts associated with marine trash and debris and their responsibilities for ensuring that trash and debris are not intentionally or accidentally discharged into the marine environment.

4. Airgun Surveys: BOEM and BSEE must require operators to ramp-up airgun arrays. Ramp-up procedures apply specifically to airguns, not electromechanical sources such as side-scan sonars, boomers, sparkers, chirp subbottom- profilers and single beam, swath or multibeam depth sounders that may be operating concurrently during seismic airgun surveys. Protocols for initiating ramp-up procedures during all seismic airgun survey operations, including airgun testing, are as follows:
   a. Operators must visually monitor the exclusion zone and adjacent waters for the absence of ESA-listed marine mammals and sea turtles for at least 60 minutes before initiating ramp-up procedures. If no ESA-listed marine mammals or sea turtles are detected, ramp-up procedures may commence. Ramp-up at night or when the exclusion zone cannot be visually monitored is prohibited if the minimum source level drops below 160 dB re 1 μPa-m (rms) (see measure iv).
      i. Initiate ramp-up procedures by firing a single airgun. The preferred airgun to begin with should be the smallest airgun, in terms of energy output (dB) and volume (in.^{3}).
      ii. Continue ramp-up by gradually activating additional airguns over a period of at least 20 min, but no longer than 40 min, until the desired operating level of the airgun array is obtained.
      iii. Immediately shut down all airguns, ceasing seismic operations at any time an ESA-listed marine mammal or sea turtle is detected entering or within the exclusion zone. After a shutdown, ramp-up and seismic operations may recommence only when the exclusion zone has been visually inspected for at least 60 minutes to ensure the absence of ESA-listed marine mammals and sea turtles.
      iv. Array source levels may be reduced using the same shot interval as that used during the seismic airgun survey, to maintain a minimum source level of 160 dB re 1 μPa-m (rms) for the duration of certain activities. The 60-minute visual clearance of the exclusion zone before ramp-up to full power is not required if the minimum source level of 160 dB re 1 μPa-m (rms) is maintained. Activities that are appropriate for maintaining the minimum source level are (1) all turns between transect lines, when a survey using the full array is being conducted immediately prior to the turn and will be resumed immediately after the turn; and (2) unscheduled, unavoidable maintenance of the airgun array that requires the interruption of a survey to

shut down the array. The survey should be resumed immediately after the repairs are completed, but should not exceed 20 minutes. Use of the minimum source level to avoid the 60-minute visual clearance of the exclusion zone is only for events that occur during a survey using the full power array. The minimum sound source level is not to be used to allow a later ramp-up after dark or in conditions when ramp-up would not otherwise be allowed.

5. Airgun Surveys: BOEM and BSEE must require operators to establish Exclusion Zones for airgun seismic surveys. The exclusion zone is the area at and below the sea surface within a radius within which animals could be exposed to a received SPL of 180 dB re 1 µPa. The 180 dB re 1 µPa distance is calculated from the center of an airgun array and validated through field verification or modeling. The radius of the exclusion zone must be calculated independently for each survey based on the configuration of the airgun array and the ambient acoustic environment, but must not be less than 500 m (1640 ft). This exclusion zone applies specifically to airguns, not electromechanical sources such as side-scan sonars, boomers, sparkers and chirp subbottom-profilers, and single beam or multibeam depth sounders that may be operating concurrently during seismic airgun surveys.

6. All Surveys: BOEM and BSEE must require operators to visually monitor for ESA-listed marine mammals and sea turtles. For surveys that require exclusion zones protected species observers must monitor for ESA-listed marine mammals and sea turtles within the survey exclusion zone. The main tasks of protected species observers are to monitor the exclusion zone for protected species and to observe and document their presence and behavior. Observers search the area around the vessel using hand-held reticle binoculars, and the unaided eye. If BOEM authorizes nighttime operations or if operations continue during periods of reduced visibility, operators must monitor the waters around the exclusion zone using available, effective monitoring technologies that may include, shipboard lighting, enhanced vision equipment, night-vision equipment and/or passive acoustic monitoring (See Passive Acoustic Monitoring below).
7. All surveys: Protected species observers who possess extensive, relevant experience as protected species observers and/or have completed a protected species observer training program as described below, or similar to the program described below, are required on all vessels conducting seismic and HRG operations that require exclusion zones.
8. Airgun Surveys: At least two protected species visual observers will be required onboard seismic airgun vessels, and undertake watches at all times during daylight hours (dawn to dusk) when

operations are being conducted, unless conditions (fog, rain, darkness) make sea surface observations impossible. If conditions deteriorate during daylight hours such that the sea surface observations are halted, visual observations must resume as soon as conditions permit. Seismic airgun operations may continue as long as minimum source levels are maintained. If ramp-up is required then seismic airgun operations cannot commence until the full exclusion zone can be monitored unless the vessel is utilizing PAM (See PAM section below).

9. Airgun Surveys: Operators must engage trained third party observers. During these observations, the following guidelines shall be followed: (a) other than brief alerts to bridge personnel of maritime hazards, no additional duties may be assigned to the observer during his/her visual observation watch (if conditions warrant more vigilant look-outs when navigating around or near maritime hazards, additional personnel must be used to ensure that watching for protected species remains the primary focus of the on-watch observers); (b) no observer will be allowed more than 4 consecutive hours on watch as a visual observer; (c) a “break” time of no less than 2 hours must be allowed, after a 4 hour watch rotation, before an observer begins another visual monitoring watch rotation (break time means no other assigned duties); and (d) no person on watch as a visual observer will be assigned a combined watch schedule of more than 12 hrs in a 24-hr period.
10. All Surveys: When BOEM requires the use of a PAM system, a qualified operator will be assigned to monitor that station and communicate with the observer on the visual observing platform. Data are recorded on paper sheets and/or a laptop computer that has direct input from the vessel’s GPS navigation system. Observers rotate among the duty stations at regular intervals, and alternate work and rest periods. In the event an ESA-listed marine mammal or sea turtle is sighted or otherwise detected within the exclusion zone, operations are suspended until the animal leaves the area and at least 60 minutes have passed, with no ESA-listed marine mammals or turtles having been observed within the exclusion zone.

Guidance for Vessel Strike Avoidance

BOEM shall condition the permits or leases it issues as part of the proposed action to require all lessees and/or operators to abide by vessel strike avoidance measures for all shipboard surveys as specified below. The guidance would be similar to Joint BOEM-BSEE NTL 2012-G01 (*Vessel Strike Avoidance and Injured/Dead Protected Species Reporting*) (USDOI, BOEM and BSEE, 2012b). These measures are required for lessees and/or operators except under extraordinary circumstances when the safety of the vessel or crew in doubt or the safety of life at sea is in question:

1. The lessee and/or operator must ensure that vessel operators and crews maintain a vigilant watch for ESA-listed whales and sea turtles and slow down or stop their vessel to avoid striking protected species.
2. The lessee and/or operators must ensure that all vessel operators must comply with 10 knot (18.5 km/h) speed restrictions in any Dynamic Management Area (DMA). In addition, the lessee and/or operator must ensure that all vessels, operate at speeds of 10 knots (18.5 km/h) or less in the mid-Atlantic Seasonal Management Areas (SMAs) from November 1 through April 30, and in the southeast SMA from November 15 to April 15 (See 50 CFR 224.105 for specific locations of SMAs).
3. For North Atlantic right whales:
    a. The lessee and/or operator must ensure all vessels maintain a separation distance of 500 m (1,640 ft) or greater from any sighted North Atlantic right whale(s).
    b. The lessee and/or operator must ensure that the following avoidance measures are taken if a vessel comes within 500 m (1,640 ft) of a right whale(s):
        i. The lessee and/or operator must ensure that while underway, any vessel must steer a course away from the right whale(s) at 10 knots (18.5 km/h) or less until the minimum separation distance has been established (unless (ii) below applies).
        ii. The lessee and/or operator must ensure that when a North Atlantic right whale is sighted in a vessel's path, or within 100 m (328 ft) of an underway vessel, the underway vessel must reduce speed and shift the engine to neutral. The lessee and/or operator must not engage the engines until the right whale(s) has moved outside of the vessel's path and beyond 100 m (328 ft).
        iii. The lessee and/or operator must ensure that if a vessel is stationary, the vessel must not engage engines until the North Atlantic right whale(s) has moved beyond 100 m (328 ft), at which time refer to point 3(b)(i).
4. ESA-listed whales other than North Atlantic right whales
    a. The lessee and/or operator must ensure all vessels maintain a separation distance of 100 m (328 ft) or greater from any sighted ESA-Listed whale (s):
    b. The lessee and/or operator must ensure that the following avoidance measures are taken if a vessel comes within 100 m (328 ft) of an ESA-Listed whale (s):
        i. The lessee and/or operator must ensure that if an ESA-Listed whale (s) (other than a North Atlantic right whale) is sighted, the vessel underway must reduce speed and shift the engine to neutral, and must not engage the engines until the

        whale(s) has moved outside of the vessel’s path and the minimum separation distance has been established.
    ii. The lessee and/or operator must ensure that if a vessel is stationary, the vessel must not engage engines until the ESA-Listed whale (s) has moved out of the vessel’s path and beyond 100 m (328 ft).

5. Sea turtles. The lessee and/or operator must ensure all vessels maintain a separation distance of 50 m (164 ft) or greater from any sighted sea turtle.
6. The lessee and/or operator must ensure that vessel operators are briefed to ensure they are familiar with the above requirements.

Training for Seismic Airgun Surveys

BOEM and BSEE shall require all visual observers to qualify for, and complete a protected species observer training course, as per the recommendations described in NOAA Fisheries Service 2012 National Standards for a Protected Species Observer and Data Management Program: A model for Seismic Surveys (Baker et al, unpublished) that includes the requirements below. All protected species observer training programs must:

1. Furnish BSEE a course information packet that includes the name and qualifications (i.e., experience, training completed, or educational background) of the instructor(s), the course outline or syllabus, and course reference material;
2. Furnish each trainee with a document stating successful completion of the course; and
3. Provide BSEE with names, affiliations, course completion dates for trainees.

The training course must include the following topics:

- i. Brief overview of the MMPA and the ESA as they relate to seismic acquisition and protection of marine mammals and sea turtles in the Atlantic Ocean.
- ii. Brief overview of seismic acquisition operations.
- iii. Overview of seismic mitigation measures and the protected species observer program.
- iv. Discussion of the role and responsibilities of the protected species observer, including:
    - a. Legal requirements (why you are here and what you do);
    - b. Professional behavior (code of conduct);
    - c. Integrity;

d. Authority of protected species observer to call for shutdown of seismic acquisition operations;
e. Assigned duties;
f. What can be asked of the observer;
g. What cannot be asked of the observer; and
h. Reporting of violations and coercion;
i. Identification of Atlantic marine mammals and sea turtles;
j. Cues and search methods for locating marine mammals and sea turtles; and,
k. Distance determination techniques and training.
l. Data collection and reporting requirements:
    1. Forms and reports to BSEE via email on the 1st and 15th of each month; and,
    2. Marine mammal or sea turtle in exclusion zone/shutdown report within 24 hr.

Visual Monitoring Methods for Seismic Airgun Surveys

All observers on duty will look for ESA-listed marine mammals and sea turtles using the naked eye and reticle, hand-held binoculars provided by the vessel operator. The observers will stand watch in a suitable location that will not interfere with navigation or operation of the vessel and that affords the observers an optimal view of the sea surface. Observers will adjust their positions to have a 360^o view of the entire area around the vessel, where possible, but ensure complete coverage of the exclusion zone.

1. Visual monitoring will begin no less than 60 minutes prior to the beginning of airgun ramp-up and continue until operations cease or sighting conditions do not allow observation of the sea surface (e.g., fog, rain, darkness).
2. If an ESA-listed marine mammal or sea turtle is observed, the observer should note and monitor the position (including latitude/longitude of the vessel and relative bearing and estimated distance to the animal) until the animal dives or moves out of visual range of the observer. Continue to observe for additional animals that may surface in the area, as often there are numerous animals that may surface at varying time intervals.
3. At any time an ESA-listed marine mammal or sea turtle is observed within the exclusion zone, whether due to the animal’s movement, the vessel’s movement, or because the animal surfaced inside the exclusion zone, the observer will call for the immediate shutdown of the operation,

including airgun firing (the vessel may continue on its course but all airgun discharges must cease). The vessel operator must comply immediately with such a call by an on-watch visual observer. Any disagreement or discussion should occur only after shutdown.

4. After a shutdown, when no ESA-listed marine mammals or sea turtles are sighted for at least a 60-minute period, airgun ramp-up of the source array may begin. Ramp-up cannot begin unless conditions allow the sea surface to be visually inspected for ESA-listed marine mammals and sea turtles for 60 minutes prior to commencement of ramp-up. Ramp-up cannot begin after dark or in conditions that prohibit visual inspection (e.g., fog, rain, etc.) of the exclusion zone unless PAM is utilized.
5. Any shutdown due to an ESA-listed marine mammal or sea turtle sighting within the exclusion zone must be followed by a 60-minute all-clear period and then a standard, full ramp-up. Any shutdown for other reasons, including, but not limited to, mechanical or electronic failure, resulting in the cessation of the sound source for a period greater than 20 minutes, must also be followed by full ramp-up procedures.
6. Periods of airgun silence not exceeding 20 minutes in duration will not require ramp-up for the resumption of seismic operations if (a) visual surveys are continued diligently throughout the silent period (requiring daylight and reasonable sighting conditions), and (b) no ESA-listed marine mammals or sea turtles are observed in the exclusion zone. If ESA-listed marine mammals or sea turtles are observed in the exclusion zone during the short silent period, resumption of seismic survey operations must be preceded by ramp-up only after no ESA-listed marine mammals or turtles have been observed in the exclusion zone for a period of 60 minutes.

Airgun Shutdown Requirements

The seismic airgun survey protocol will require shutdown of the airgun array any time an ESA-listed marine mammal or sea turtle is observed within the exclusion zone, whether due to the animal’s movement, the vessel’s movement, or because the animal surfaced inside the exclusion zone. In the event of a shutdown, seismic operations and ramp-up of airguns would resume only when the sighted animal has cleared the exclusion zone and no other ESA-listed marine mammals or sea turtles have been sighted within the exclusion zone for at least 60 minutes.

Passive Acoustic Monitoring for Airgun Seismic Surveys

BOEM shall condition the permits or leases it issues as part of the proposed action to require operators to include passive acoustic monitoring (PAM) as part of the protected species observer program for airgun seismic surveys. Operators using passive acoustic array devices are required to employ an observer proficient in passive acoustic monitoring operations and methods to detect vocalizing cetaceans. By using PAM devices, operators can ramp-up and start/resume a seismic survey during times of reduced visibility (e.g., darkness, fog, rain, etc.) when such ramp-up otherwise would not be permitted using only visual observers. All other mitigation measures required for seismic surveys remain applicable to seismic survey operations. A description of the passive acoustic system, the software used, and the monitoring plan should also be reported to BSEE and NMFS prior to its use. NMFS will work with operators to determine which PAM systems are adequate for use.

Non-Airgun HRG Survey Protocol for Renewable Energy and Marine Minerals Sites

The HRG surveys for renewable energy and marine minerals sites would use only electromechanical sources such as side-scan sonar, boomer and chirp subbottom profilers, and single beam, swath and multibeam depth sounders. All authorizations for non-airgun HRG surveys would include the requirements listed for all surveys under the General Monitoring and Mitigation Requirements section and the Guidance for Vessel Strike Avoidance sections as applicable above. All authorizations for non-airgun HRG surveys using sound sources below 200 kHz would include requirements for visual monitoring of an exclusion zone by protected species observers and start-up and shut-down requirements as described below. The HRG surveys for oil and gas exploration typically, but not always, use electromechanical sources operating concurrently with airgun arrays. If seismic airguns are used, these surveys would be subject to the seismic airgun survey protocol described above.

Non-Airgun HRG Protocol Requirements

1. All non-airgun HRG surveys must comply with the requirements for vessel strike avoidance (See Guidance for Vessel Strike Avoidance above). The recommended separation distance for North Atlantic right whales of 500 m (1,640 ft) would remain in effect during HRG surveys since it exceeds the exclusion zone radius specified below. Recommended separation distances for other ESA-listed whales are less than the exclusion zone radius of 200 m (656 ft) required for these species and, therefore, would be superseded by the exclusion zone radius during HRG surveys.
2. All non-airgun HRG surveys within Southeast Right Whale Critical Habitat during the calving and nursing season shall operate sound sources at frequencies above 30 kHz except for surveys meeting critical program needs. Surveys BOEM determines to be critical to program needs can be authorized to use frequencies at and below 30 kHz (within the audibility range of North Atlantic right whales). This determination will consider whether

survey planning could have scheduled survey activities outside of the calving and nursing season and how the particular survey fills a critical need of the program. For these surveys, BOEM will only authorize the use of sound sources at and below 30 kHz during daylight hours (Marine Minerals Program) or use of sound sources above 200 kHz at night and sound sources at and below 30 kHz and within the audibility range of North Atlantic right whales during daylight hours (Renewable Energy Program). Only 1-2 surveys that may require the use of sound sources at and below 30 kHz are anticipated to occur per year within North Atlantic right whale critical habitat in the southeast during the calving and nursing season.

3. All non-airgun HRG surveys within or adjacent to North Atlantic right whale critical habitat in the southeastern United States, within or adjacent to SMAs or within established DMAs, BOEM and BSEE must require operators to implement the following measures :
    a. Vessel Strike Avoidance: A distance of 500 m (1,640 ft) from right whales must be maintained during transit and the Early Warning System, Sighting Advisory System, and Mandatory Ship Reporting System data notifying mariners of right whale presence must be monitored during transit and operation.
    b. A 200-m (656 ft) radius exclusion zone would be monitored by a protected species observer that would encompass the 180 dB re 1 μPa·m$_{rms}$ isopleth of any acoustic source. If the source produces a wider radius than this, than the exclusion zone would be increased to the field verification or modeled distance.
    c. Non-airgun HRG transmissions within a DMA must cease within 24 hours of the DMA's establishment.
    d. At least one protected species observer would be required on watch aboard HRG survey vessels at all times during daylight hours (dawn to dusk – i.e., from about 30 minutes before sunrise to 30 minutes after sunset) when survey operations are being conducted, unless conditions (fog, rain, fading light) make sea surface observations impossible. If conditions deteriorate during daylight hours such that the sea surface observations are halted, visual observations must resume as soon as conditions permit. Ongoing activities may continue but may not be initiated under such conditions (i.e., without appropriate pre-activity monitoring), unless as provided for under General Mitigation and Monitoring Requirements #6.

    i. Visual monitoring will begin no less than 60 minutes prior to start-up and continue until operations cease or sighting conditions do not allow observation of the sea surface (e.g., fog, rain, fading light).

    ii. At any time an ESA-listed marine mammal or sea turtle is observed within the exclusion zone, whether due to the animal's movement, the vessel's movement, or because the animal surfaced inside the exclusion zone, the observer will call for the immediate shutdown of the operation. The vessel operator must comply immediately with such a call by an on-watch visual observer. Any disagreement or discussion should occur only after shutdown.

The following additional requirements apply to non-airgun HRG surveys in which one or more active acoustic sound sources will be operating at frequencies less than 200 kHz.

1. A 200-m (656-ft) radius exclusion zone will be established and monitored around the sound source. However, if this exclusion zone does not encompass the 180-dB radius, as validated through field verification or modeling, for any acoustic source proposed for use, BOEM may authorize, with NMFS' concurrence, surveys having a larger exclusion zone to encompass the 180-dB radius. Effectiveness can be evidenced by monitoring studies that have been carried out (i.e., Barkaszi et al 2012). Demonstration may also be in the class of boat used for the survey work, some of which may not have flying bridges or an observer height on deck that easily allows visual observation of the larger exclusion zone.
2. At least one protected species observer would be required on watch aboard HRG survey vessels at all times during daylight hours (dawn to dusk – i.e., from about 30 minutes before sunrise to 30 minutes after sunset) when survey operations are being conducted, unless conditions (fog, rain, fading light) make sea surface observations impossible. If conditions deteriorate during daylight hours such that the sea surface observations are halted, visual observations must resume as soon as conditions permit. Ongoing activities may continue but may not be initiated under such conditions (i.e., without appropriate pre-activity monitoring), unless as provided for under General Mitigation and Monitoring Requirements #6.
3. Active acoustic sound sources must not be activated until the protected species observer has reported the exclusion zone clear of all ESA-listed marine mammals and sea turtles for 60 minutes.

4. If any ESA-listed marine mammal or sea turtle is sighted at or within the exclusion zone, operators are required to immediately shutdown the equipment. Subsequent restart of the equipment may only occur following a confirmation that the exclusion zone if clear of ESA-listed marine mammals and sea turtles for 60 minutes.

Borehole Seismic Surveys

Borehole seismic surveys (completed during VSP surveys) differ from surface seismic surveys in a number of ways, including the use of much smaller airgun arrays, having an average survey time of 12-24 hours, utilizing a sound source that may or may not move at 7.4-9.3 km/hr (4-5 kn), and requiring the capability of moving the receiver in the borehole between shots. Due to these differences, the following altered mitigation measures apply only to borehole seismic surveys:

1. During daylight hours, when visual observations of the exclusion zone are being performed as required in this protocol, borehole seismic operations will not be required to ramp-up for shutdowns of 30 min or less in duration, as long as no ESA-listed marine mammals or sea turtles are observed in the exclusion zone during the shutdown. If an ESA-listed marine mammal or sea turtle is sighted in the exclusion zone, ramp-up is required and may begin only after visual surveys confirm that the exclusion zone has been clear for 60 minutes.
2. Nighttime or poor visibility ramp-up is allowed only when passive acoustics are used to ensure that no ESA-listed marine mammals are present in the exclusion zone (as for all other seismic surveys). Operators are strongly encouraged to acquire the survey in daylight hours when possible.
3. Protected species observers must be used during daylight hours, as required in this protocol, and may be stationed either on the source boat or on the associated drilling rig or platform if a clear view of the exclusion zone and adjacent waters is available.
4. All other mitigations and provisions for seismic surveys as set forth in this protocol will apply to borehole seismic surveys.

Reporting Requirements for Seismic Airgun Surveys

BOEM and BSEE shall require operators to submit observer effort reports and survey reports for each seismic airgun survey conducted. In order to accommodate various vessels’ bridge practices and preferences, vessel operators and observers may design data reporting forms in whatever format they deem convenient and appropriate. Alternatively, observers or vessel operators may adopt the United

Kingdom's JNCC forms (available at their website, http://www.jncc.gov.uk). At a minimum, the following items should be recorded and included in reports to the BOEM and BSEE (see NOAA Fisheries Service 2012 National Standards for a Protected Species Observer and Data Management Program: A model for Seismic Surveys (Baker et al, unpublished).

<u>Observer Effort Report</u>: BOEM requires the submission of observer effort reports to BSEE on the 1st and the 15th of each month for each day seismic acquisition operations are conducted. These reports must include:

1. Vessel name;
2. Observers' names and affiliations;
3. Survey type (e.g., site, 3D, 4D);
4. BOEM Permit Number (for "off-lease seismic surveys") or OCS Lease Number (for "on-lease seismic surveys");
5. Date;
6. Time and latitude/longitude when daily visual survey began;
7. Time and latitude/longitude when daily visual survey ended; and
8. Average environmental conditions while on each visual survey rotation and session as well as when any conditions change during the rotation ,each session, including:
    a. Wind speed and direction;
    b. Sea state (glassy, slight, choppy, rough, or Beaufort scale);
    c. Swell (low, medium, high, or swell height in meters); and
    d. Overall visibility (poor, moderate, good).

<u>Survey Report</u>: BOEM requires the submission of survey reports to BSEE on the 1st and the 15th of the month for each day seismic acquisition operations are conducted and airguns are discharged. These reports must include:

1. Vessel name;
2. Survey type (e.g., site, 3D, 4D);
3. BOEM Permit Number (for "off-lease seismic surveys") or OCS Lease Number (for "on-lease seismic surveys"), if applicable;
4. Date;
5. Time pre-ramp-up survey begins;
6. Observations of marine mammals and sea turtles seen during pre-ramp-up surveys
7. Time ramp-up begins;

8. Observations of marine mammals and sea turtles seen during ramp-up;
9. Time sound source (airguns or HRG equipment) is operating at the desired intensity;
10. Observations of marine mammals and sea turtles seen during surveys;
11. If marine mammals or sea turtles were seen, was any action taken (i.e., survey delayed, guns shut down)?
12. Reason that marine mammals and sea turtles might not have been observed (e.g., swell, glare, fog); and
13. Time sound source (airgun array or HRG equipment) stops firing.

Sighting Report: BOEM shall require the submission of reports to BSEE for marine mammals and sea turtles sighted during seismic and HRG surveys on the 1st and the 15th of each month except as indicated below. These reports are in addition to any reports required as a condition of the geophysical permit and must include:

1. Vessel name;
2. Survey type (e.g., site, 3D, 4D);
3. BOEM Permit Number (for "off-lease seismic surveys") or OCS Lease Number (for "on-lease seismic surveys");
4. Date;
5. Time;
6. Watch status (Were you on watch or was this sighting made opportunistically by you or someone else?);
7. Observer or person who made the sighting;
8. Latitude/longitude of vessel;
9. Bearing of vessel; (true compass direction);
10. Bearing (true compass direction) and estimated range to animal(s) at first sighting;
11. Water depth (meters);
12. Species (or identification to lowest possible taxonomic level);
13. Certainty of identification (sure, most likely, best guess);
14. Total number of animals;
15. Number of juveniles;
16. Description (as many distinguishing features as possible of each individual seen, including length, shape, color and pattern, scars or marks, shape and size of dorsal fin, shape of head, and blow characteristics);
17. Direction of animal's travel – compass direction;

18. Direction of animal's travel – related to the vessel (drawing preferably);
19. Behavior (as explicit and detailed as possible; note any observed changes in behavior);
20. Activity of vessel;
21. Airguns firing? (yes or no); and
22. Closest distance (meters) to animals from center of airgun or airgun array (whether firing or not).

BOEM shall also require operators to report sightings of ESA-listed marine mammals or sea turtles within the exclusion zone that resulted in a shutdown of the airguns within 24 hours of the shut-down to BSEE. The sighting report must include the observed behavior of the animal(s) before shutdown, the observed behavior following shutdown (specifically noting any change in behavior), and the length of time between shutdown and subsequent ramp-up to resume the seismic survey (note if seismic survey was not resumed as soon as possible following shutdown). These sightings should also be included in the first regular semi-monthly report following the incident. Sighting forms should be scanned (or data typed) and sent via email to the BOEM and BSEE. If passive acoustic monitoring is used, an assessment must be included of the usefulness, effectiveness, and problems encountered with the use of that method of marine mammal detection in the reports described in this protocol.

Non Airgun HRG and Geotechnical Survey Reporting Requirements

Protected Species Observer Reports: Data on all protected species observations must be recorded by the protected species observer based on standard marine mammal observer data collection protocols. This information must include:

1. Vessel name;
2. Observers' names, affiliations and resumes;
3. Date;
4. Time and latitude/longitude when daily visual survey began;
5. Time and latitude/longitude when daily visual survey ended; and
6. Average environmental conditions during visual surveys including:
   a. Wind speed and direction;
   b. Sea state (glassy, slight, choppy, rough, or Beaufort scale);
   c. Swell (low, medium, high, or swell height in meters); and
   d. Overall visibility (poor, moderate, good).

7. Species (or identification to lowest possible taxonomic level);
8. Certainty of identification (sure, most likely, best guess);
9. Total number of animals;
10. Number of calves, and juveniles (if distinguishable);
11. Description (as many distinguishing features as possible of each individual seen, including length, shape, color and pattern, scars or marks, shape and size of dorsal fin, shape of head, and blow characteristics);
12. Direction of animal's travel – related to the vessel (drawing preferably);
13. Behavior (as explicit and detailed as possible; note any observed changes in behavior);
14. Activity of vessel when sighting occurred.

Non-Airgun and Geotechnical Reporting Requirements: The lessee/operator must provide BOEM with a report within ninety (90) calendar days following the commencement of HRG and/or geotechnical sampling activities that includes all protected species observer reports, a summary of the survey activities and an estimate of the number of listed marine mammals and sea turtles observed or taken during these survey activities.

General Reporting to NMFS

1. BOEM and/or BSEE shall deliver a report of data collected during the first year and every year, thereafter delivered in March of the following year for which survey activities are conducted. These reports must include the information contained in Observer Reports, Survey Reports and the Sighting Reports submitted by all survey operators (Seismic Airgun, Non-airgun HRG and Geotechnical). These reports shall also include the results, if any, of coordination with coastal marine mammal stranding networks. The annual report must address all affected threatened and endangered species as well as critical habitat (including the number and durations of surveys authorized) addressed in this Opinion. The annual reports shall be submitted to: (1) Chief, Endangered Species Act Interagency Cooperation Division, 1315 East-West Highway, Silver Spring, MD 20910.

2. BOEM and BSEE shall deliver a 4-year report (covering the years 2013-2016) by September 2017 and a final comprehensive report (covering the years 2013-2020) by September 2021, to the Chief, Endangered Species Act Interagency Cooperation Division, Office of Protected Resources, containing the same information cited in condition 1 of this section. These reports will also address the effectiveness of all mitigation and monitoring, including PAM systems. This report will also address any conservation recommendations BOEM and/or BSEE have adopted and their progress to date.

3. In the unanticipated event that the specified activity clearly causes the take of an ESA-listed marine mammal or sea turtle in a manner not exempted by this Incidental Take Statement, such as serious injury or mortality (e.g., ship-strike, gear interaction, and/or entanglement), survey operators shall immediately report the incident to BOEM and BSEE and the Chief, Endangered, Species Act Interagency Cooperation Division, Office of Protected Resources, NMFS, at 301-427-8403 and/or by email to kellie.foster-taylor@noaa.gov and the appropriate NMFS Regional Stranding Coordinator (NMFS Northeast Regional Stranding Coordinator for the mid-Atlantic Planning Area and NMFS Southeast Regional Stranding Coordinator for South Atlantic Planning Area). The report must include the following information:

    (a) 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 hours preceding the incident; water depth; environmental conditions (e.g., wind speed and direction, Beaufort sea state, cloud cover, and visibility); description of marine mammal and sea turtle observations in the 24 hours 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).

Following the submittal of this incident report, BOEM and BSEE should contact NMFS to determine whether the incident requires reinitiation of formal consultation. NMFS shall work with BOEM/BSEE and the operator to determine what is necessary to minimize the likelihood of further prohibited take. Activities may continue while NMFS reviews the circumstances of the incident. NMFS will work with BOEM/BSEE and the operator to determine whether modifications in the activities are appropriate.

In the event that an operator discovers an injured or dead marine mammal, and the lead PSO determines that the cause of the injury or death is unknown and the death is relatively recent (i.e., in less than a moderate state of decomposition as described in the next paragraph), BOEM and/or BSEE shall report the incident to the Chief, Endangered Species Act Interagency Cooperation Division, Office of Protected Resources, NMFS, at 301-427-8403, and/or by email to kellie.foster-taylor@noaa.gov, and to the appropriate NMFS Stranding Coordinator (Northeast Regional Office for surveys in the mid-Atlantic Planning Area, or Southeast Regional Office for surveys in the South Atlantic Planning Area) within 24 hours of the discovery. Photographs or video footage (if available) or other documentation of the stranded animal sighting should be sent to NMFS and the Marine Mammal Stranding Network. The report must include the same information identified in Condition 3(a) of this section. Activities may continue while NMFS reviews the circumstances of the incident. NMFS will work with BOEM/BSEE and the operator to determine whether modifications in the activities are appropriate.

In the event that an operator discovers an injured or dead marine mammal, and the lead PSO determines that the injury or death is not associated with or related to the activities authorized in ITS (e.g., previously wounded animal, carcass with moderate to advanced decomposition, or scavenger damage), BOEM and/or BSEE shall report the incident to the Chief, Endangered Species Act Interagency Cooperation Division, Office of Protected Resources, NMFS, at 301-427-8403, and/or by email to kellie.foster-taylor@noaa.gov, and to the appropriate NMFS Stranding Coordinator (Northeast Regional Office for surveys in the mid-Atlantic Planning Area, or Southeast Regional Office for surveys in the South Atlantic Planning Area) within 24 hours of the discovery. Photographs or video footage (if available) or other documentation of the stranded animal sighting should be sent to NMFS and the Marine Mammal Stranding Network. Activities may continue while NMFS reviews the circumstances of the incident.

CONSERVATION RECOMMENDATIONS

Section 7(a)(1) of the ESA directs federal agencies to use their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of endangered and threatened species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on listed species or critical habitat, to help implement recovery plans, or to develop information. We recommend the following conservation recommendations, which would provide information for future consultations involving G&G activities that may affect endangered or threatened species:

1. BOEM and BSEE should work with NMFS Endangered Species Act Interagency Cooperation Division and other relevant stakeholders (the Marine Mammal Commission, International Whaling Commission, and the marine mammal research community) to develop a method for assessing the cumulative impacts of anthropogenic noise on cetaceans, pinnipeds, sea turtles, and other marine animals. This includes the cumulative impacts on the distribution, abundance, and the physiological, behavioral and social ecology of these species.
2. BOEM and BSEE should review reports submitted for seismic and HRG surveys and compile and analyze information to improve agency estimates of the number of the different species of marine mammals and sea turtles that are likely to be exposed to sounds from seismic surveys, the response of those species to this exposure, and the probable consequences of those responses on the life history of individual animals. The results should be provided to the Endangered Species Act Interagency Cooperation Division as part of requests for consultation on future proposals to authorize incidental harassment.
3. To the maximum extent practicable, BOEM and BSEE should encourage operators to schedule seismic operations (i.e., shooting airguns) during daylight hours and deploy/retrieve equipment to nighttime hours.
4. To the maximum extent practicable, BOEM should require survey operators to plan survey tracklines (especially when near land) starting from the coast (inshore) and proceeding towards the sea (offshore) to avoid trapping marine mammals in shallow water.
5. BOEM and BSEE should monitor ambient sound levels within the Action Area as well as elevated sound levels due to seismic and HRG activities to gain insight into the incremental impact of increasing sound levels on listed species in the Northwest Atlantic.
6. BOEM and BSEE should partner with other entities that conduct seismic and sonar activities such as the U.S. Navy and the National Science Foundation to coordinate the timing and spatial arrangement of surveys and training exercises.

7. BOEM and BSEE should also partner with the National Science Foundation and other agencies as opportunities arise to explore sharing *in situ* propagation measurements used for marine mammal exposure models.

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

REINITIATION NOTICE

This concludes formal consultation on BOEM's and BSEE's proposed geological and geophysical activities in the Mid- and South Atlantic Planning Areas. As provided in 50 CFR §402.16, reinitiation of consultation is required where control over the action has been retained (or is authorized by law) and if: (1) the amount or extent of incidental take is exceeded; (2) new information reveals effects of the agency action that may affect listed species or critical habitat in a manner or to an extent not considered in this opinion; (3) the agency action is subsequently modified in a manner that causes an effect to the listed species or critical habitat not considered in this opinion; or (4) a new species is listed or critical habitat designated that may be affected by the action. In instances where the amount or extent of authorized take is exceeded, section 7 consultation must be reinitiated immediately.

The analyses that form the basis for the conclusions of this biological opinion are based on estimates of survey effort specified in applications BOEM submitted previously for seismic surveys along the eastern seaboard of the United States. Similarly, the estimates of the number of listed species that are likely to be "taken" as a result of the proposed seismic surveys are based on those same estimates.

Because of the limited number of samples, there is considerable uncertainty associated with those "take" estimates, particularly with estimates of the number of surveys that would actually occur, the total number of hours active sources associated with those surveys would operate, and where the surveys would occur. Estimates associated with future surveys, when combined with existing estimates, would increase the sample size as well as the level of confidence associated with any new estimates we generate from the accumulated data. Based on our analyses, data from 12 additional seismic surveys would provide a sample size large enough to make us more confident of the data on which the conclusions of this opinion are based. If the mean values of new estimates based on those 12 additional surveys exceed the mean values of the estimates contained in this opinion (for example, estimates of total number of operating hours, operating hours associated with different kinds of survey equipment, spatial distribution of operating hours, and variability in survey effort) by 25% or more (we estimated mean values and confidence intervals based on the t-distribution and using the procedures and formulae described by Cumming (2012)), we would treat that new data as significant new information that reveals effects we have not considered in this opinion. Those new data would also be evidence that the amount or extent of incidental take would have been exceeded. In either case, reinitiation of formal consultation would be required.

Similarly, existing density estimates of large whales are the basis for the exposure estimates contained in this opinion and for the take estimates contained in the Incidental Take Statement of this opinion. New density estimates are expected in the near future through the Cetacean Sound Mapping Program. Any new density estimates may also constitute significant new information that would require reinitiation of consultation.

Literature Cited

Abgrall, P., V.D. Moulton and W.J. Richardson. 2008. Updated review of scientific information on impacts of seismic survey sound on marine mammals, 2004-present. LGL Rep. SA973-1. Rep. from LGL Limited, St. John's, NL and King City, ON, for Department of Fisheries and Oceans, Habitat Science Branch, Ottawa, ON. 27 p. + appendices

Acevedo-Gutierrez, A., D. A. Croll, and B. R. Tershy. 2002. High feeding costs limit dive time in the largest whales. The journal of Experimental Biology 205:1747-1753.

Acevedo-Guttierrez, A., B. Brennan, P. Rodriguez, and M. Thomas. 1997. Resightings and behavior of false killer whales (*Pseudorca crassidens*) in Costa Rica. Marine Mammal Science 13:7.

Agler, B. A., R. L. Schooley, S. E. Frohock, S. K. Katona, and I. E. Seipt. 1993. Reproduction of photographically identified fin whales, *Balaenoptera physalus*, from the Gulf of Maine. Journal of Mammalogy 74:577-587.

Aguilar, A. 2000. Population biology, conservation threats and status of Mediterranean striped dolphins (*Stenella coeruleoalba*). Journal of Cetacean Research and Management 2.

Aguilar, A. and A. Borrell. 1990. Patterns of lipid content and stratification in the blubber of fin whales (*Balaenoptera physalus*). Journal of Mammalogy 71:544-554.

Aguilar, R., J. Mas and X. Pastor (1995). Impact of Spanish swordfish longline fisheries on the loggerhead sea turtle, Caretta caretta, population in the western Mediterranean. 12th Annual Workshop on Sea Turtle Biology and Conservation, Jekyll Island, Georgia.

Akamatsu, T., T. Okumura, N. Novarini, and H. Y. Yan. 2002. Empirical refinements applicable to the recording of fish sounds in small tanks. The Journal of the Acoustical Society of America 112:10.

Akamatsu, T., D. Wang, K. Nakamura, and K. Wang. 1998. Echolocation range of captive and free-ranging baiji (*Lipotes vexillifer*), finless porpoise (*Neophocaena phocaenoides*), and bottlenose dolphin (*Tursiops truncatus*). The Journal of the Acoustical Society of America 104:2511.

Alava, J. J., M. J. Barragan, C. Castro, and R. Carvajal. 2005. A note on strandings and entanglements of humpback whales (*Megaptera novaeangliae*) in Ecuador. Journal of Cetacean Research and Management 7:163 - 168.

Alfano, G., A. D. Maio, and A. Farina. 2004. Model-based adaptive detection of range-spread targets. IEE Proceedings, Radar, Sonar, and Navigation 151:9.

Alfaro-Shigueto J, P.H. Dutton, J. Mangel, and D. Vega. 2004. First confirmed occurrence of loggerhead turtles in Peru. Marine Turtle Newsletter 103:7-11.

Allen, B. M., and R. P. Angliss. 2012. Alaska marine mammal stock assessments, 2011. U.S. Dep. Commer., NOAA Tech. Memo. NMFS-AFSC-234, 288 p.

Allen, R. C. and I. A. N. Keay. 2004. Saving the whales: lessons from the extinction of the eastern Arctic bowhead. The Journal of Economic History 64:400 - 432.

Alonso, M. K., S. N. Pedraza, A. C. M. Schiavini, R. N. P. Goodall, and E. A. Crespo. 1999. Stomach contents of false killer whales (*Pseudorca crassidens*) stranded on the coasts of the Strait of Magellan, Tierra del Fuego. Marine Mammal Science 15:13.

Amano, M. and M. Yoshioka. 2003. Sperm whale diving behavior monitored using a suction-cup-attached TDR tag. Marine Ecology Progress Series 258:6.

Anderson, R. C. 2005. Observations of cetaceans in the Maldives, 1990-2002. Journal of Cetacean Research and Management 2.

André, M., E. Degollada, and A. Fernández. 2001. Hearing loss in long-term low frequency sounds exposed sperm whales. Pages 6-7 *in* 14th Biennial Conference on the Biology of Marine Mammals, Vancouver, Canada.

André, M. and C. Kamminga. 2000. Rhythmic dimension in the echolocation click trains of sperm whales: a possible function of identification and communication. Journal of the Marine Biological Association of the UK 80:7.

André, M., M. Terada, and Y. Watanabe. 1997. Sperm whale (*Physeter macrocephalus*) behavioral response after the playback of artificial sounds. Reports of the International Whaling Commission 47:499 - 504.

Anonymous. 1994. 94/C 300/37: Protection of the monk seal (*Monachus Monachus*). Official Journal of the European Communities. Information and Notices 37:19.

Anonymous. 1996. 96/C 161/65: The monk seal and the threat of extinction. Official Journal of the European Communities. Information and Notices 39:38.

Anonymous. 1998. 98/C 82/128: Monk seal catastrophe off the coast of Mauritania. Official Journal of the European Communities. Information and Notices 41:77

Anonymous. 2000. Report of the 1999 Annual Meeting of the scientific committee of the International whaling commission. Journal of Cetacean Research and Management 2.

Anonymous. 2001. Report of the workshop on status and trends of Western North Atlantic right whales. Journal of Cetacean Research and Management Supplement 2:61 - 87.

Anonymous. 2001. Report of the workshop on the comprehensive assessment of right whales: a worldwide comparison. Journal of Cetacean Research and Management Supplement 2:1 - 60.

Anonymous. 2002. Navy to limit sonar testing thought to hurt sea mammals. Page 25 New York Times.

Anonymous. 2002. Report of the third workshop on the development of an aboriginal subsistence whaling management procedure. Journal of Cetacean Research and Management 4.

Anonymous. 2003. Report of the sub-committee on bowhead, right and gray whales. Journal of Cetacean Research and Management 5:226-247.

Anonymous. 2003. Report of the sub-committee on the comprehensive assessment of humpback whales. Journal of Cetacean Research and Management 5: suppl., 293-323. Apr

Anonymous. 2005. Annex K. Report of the standing working group on environmental concerns. Journal of Cetacean Research and Management 7:267 - 281.

Anonymous. 2005. Report of the Scientific Committee. Journal of Cetacean Research and Management 7:411.

Antonelis, G. A., J. D. Baker, and J. J. Polovina. 2003. Improved body condition of weaned Hawaiian monk seal pups associated with El Nino events: Potential benefits to an endangered species. Marine Mammal Science 19:9.

Aono, S., K. Takagi, and H. Takinami. 1999. The effect of some amusement sounds on hearing in terms of temporary threshold shift. Noise control engineering journal 47:3.

Apple, T. C. 2001. Spatial and temporal variation of sperm whale (*Physeter macrocephalus*) codas in the northern Gulf of Mexico. The Journal of the Acoustical Society of America 109:2390.

Armstrong, J. L. and J. E. Hightower. 2002. Potential for restoration of the Roanoke River population of Atlantic sturgeon. Journal of Applied Ichthyology. 18:475-480.

Armstrong, P. S. 2001. Conservation genetics of the Hawai'ian monk seal (*Monachus schauinslandi*). Dissertation Abstracts International 62:95.

Aron, W., W. Burke, and M. t. Freeman. 2002. Scientists versus whaling: Science, advocacy, and errors of judgment. Bioscience 52:1137-1140.

Atlantic States Marine Fisheries Commission (ASMFC). 1998. Amendment 1 to the interstate fishery management plan for Atlantic sturgeon. Management Report No. 31, 43 pp.

Atlantic States Marine Fisheries Commission (ASMFC). 2007. Estimation of Atlantic Sturgeon Bycatch in Coastal Atlantic Commercial Fisheries of New England and the Mid-Atlantic. Special Report to the ASMFC Atlantic Sturgeon Management Board, August 2007.

Atlantic Sturgeon Status Review Team (ASSRT). 2007. Status Review of Atlantic sturgeon (Acipenser oxyrinchus oxyrinchus). Report to National Marine Fisheries Service, Northeast Regional Office. February 23, 2007. 174 pp.

Attias, J., S. Sapir, I. Bresloff, I. Reshef-Haran, and H. Ising. 2004. Reduction in noise-induced temporary threshold shift in humans following oral magnesium intake. Clinical Otolaryngology 29:635-641.

Au, W. and M. Green. 2000. Acoustic interaction of humpback whales and whale-watching boats. Marine Environmental Research 49:469-481.

Au, W., P. E. Nachtigall, and J. Pawloski. 1997. Acoustic effects of the ATOC signal (75 Hz, 195 dB) on dolphins and whales. The Journal of the Acoustical Society of America 101:2973-2977.

Au, W. W. L. 1997. Some hot topics in animal bioacoustics. The Journal of the Acoustical Society of America 101:10.

Au, W. W. L. 2002. Snapping shrimps: The bane of bioacoustics monitoring of coral reefs. The Journal of the Acoustical Society of America 112:2202.

Au, W. W. L., L. N. Andersen, A. R. Rasmussen, H. L. Roitblat, and P. E. Nachtigall. 1995. Neural network modeling of a dolphin's sonar discrimination capabilities. The Journal of the Acoustical Society of America 98:8.

Au, W. W. L. and K. J. Benoit-Bird. 2003. Automatic gain control in the echolocation system of dolphins. Nature 423:861-863.

Au, W. W. L., A. Frankel, D. A. Helweg, and D. H. Cato. 2001. Against the humpback whale sonar hypothesis. IEEE Journal of Oceanic Engineering 26:5.

Au, W. W. L., R. A. Kastelein, T. Rippe, and N. M. Schooneman. 1999. Transmission beam pattern and echolocation signals of a harbor porpoise (*Phocoena phocoena*). The Journal of the Acoustical Society of America 106:3699.

Au, W. W. L. and P. E. Nachtigall. 1997. Acoustics of echolocating dolphins and small whales. Marine Behaviour and Physiology 29:36.

Au, W. W. L., J. L. Pawloski, P. E. Nachtigall, M. Blonz, and R. C. Gisner. 1995. Echolocation signals and transmission beam pattern of a false killer whale (*Pseudorca crassidens*). The Journal of the Acoustical Society of America 98:51.

Au, D., and W. Perryman. 1982. Movement and speed of dolphin schools responding to an approaching ship. Fishery Bulletin 80:371-379.

Augier, H., S. Gulbasdian, and G. Ramonda. 1998. Heavy metal contents of the sperm whale *Physeter macrocephalus* and the bottle-nosed dolphin *Tursiops truncatus* beached on Languedoc-Roussillon seaside (Mediterranean, France). Toxicological and Environmental Chemistry 67:6.

Aurioles-Gamboa, D. 1992. Notes on a mass stranding of Baird's beaked whales in the Gulf of California, Mexico. California Fish and Game 78:116-123.

Bain, M. B., D. L. Peterson, K. K. Arend, and N. Haley. 1999. Atlantic Sturgeon Population Monitoring for the Hudson River Estuary: Sampling Design and Gear Recommendations. Hudson Rivers Fisheries Unit, New York State Department of Environmental Conservation, New Paltz, NY and The Hudson River Foundation, New York, NY.

Baird, R. and H. Whitehead. 2000. Social organization of mammal-eating killer whales: group stability and dispersal patterns. The Canadian Journal of Zoology 78:10.

Baird, R. W. 2000. The killer whale: foraging specializations and group hunting. Pages 127-153 *in* J. Mann, R. C. Connor, P. L. Tyack, and H. Whitehead, editors. Cetacean societies. Field studies of dolphins and whales. University of Chicago Press, Chicago, Illinois.

Baird, R. W. and L. M. Dill. 1995. Occurrence and behaviour of transient killer whales: Seasonal and pod-specific variability, foraging behaviour, and prey handling. The Canadian Journal of Zoology 73:12.

Baird, R. W., M. B. Hanson, and L. M. Dill. 2005. Factors influencing the diving behaviour of fish-eating killer whales: sex differences and diel and interannual variation in diving rates. The Canadian Journal of Zoology 83:257-267.

Baird, R. W., G. S. Schorr, D. L. Webster, S. D. Mahaffy, A. B. Douglas, A. M. Gorgone, and D. J. McSweeney. 2006. A survey for odontocete cetaceans off Kaua'i and Ni'ihau, Hawai'i, during October and November 2005: evidence for population structure and site fidelity. Unpublished report prepared by Cascadia Collective of Olympia, Washington, under Order No. AB133F05SE5197, U.S. Department of Commerce, National Marine Fisheries Service, Pacific Islands Fisheries Science Center, Honolulu, Hawaii.

Baker, A. N. and A. L. Van Helden. 1999. New records of beaked whales, Genus *Mesoplodon*, from New Zealand (Cetacea: *Ziphiidae*). Journal of the Royal Society of New Zealand 29:235.

Baker, C. S., G. M. Lento, F. Cipriano, and S. R. Palumbi. 2000. Predicted decline of protected whales based on molecular genetic monitoring of Japanese and Korean markets. Proceedings of the Royal Society of London, Series B: Biological Sciences 267:1191-1199.

Balazs, G. 2006. Pelagic research of Pacific loggerhead sea turtles in partnership with Japan and Taiwan. Pages 31-33 in Kinan, I. (compiler). Proceedings of the Second Western Pacific Sea Turtle Cooperative Research & Management Workshop. Volume II: North Pacific Loggerhead Sea Turtles. Western Pacific Regional Fishery Management Council, Honolulu, Hawaii.

Baldwin, R.M. 1992. Nesting turtles on Masirah Island: Management issues, options, and research requirements. Report, Ministry of Regional Municipalities and Environment, Oman.

Baldwin, R., G.R. Hughes, and R.I.T. Prince. 2003. Loggerhead turtles in the Indian Ocean. Pages 218-232 in Bolten, A.B. and B.E. Witherington (editors). Loggerhead Sea Turtles. Smithsonian Institution Press, Washington, D.C.

Balcomb, K. C. and D. E. Claridge. 2001. Mass whale mortality: U.S. Navy exercises cause strandings. Bahamian Journal of Science 8:1 - 12.

Ballance, L. T., R. C. Anderson, R. L. Pitman, K. Stafford, A. Shaan, Z. Waheed, and R. L. Brownell, Jr. 2001. Cetacean sightings around the Republic of the Maldives, April 1998. Journal of Cetacean Research and Management 3:213 - 218.

Barkaszi, M.J., M. Butler, R. Compton, A. Unietis, and B. Bennet. 2012. Seismic Survey Mitigation Measures and Marine Mammal Observer Reports. Gulf of Mexico Region, New Orleans, LA.

OCS Study BOEM 2012-015. 28 pp w Appendices. Available at: http://www.data.boem.gov/PI/PDFImages/ESPIS/5/5177.pdf

Barlow, J. 1994. Abundance of large whales in California coastal waters: a comparison of ship surveys in 1979/80 and in 1991. Reports of the International Whaling Commission 44:399-406.

Barlow, J. 2006. Cetacean abundance in Hawaiian waters estimated from a summer/fall survey in 2002. Marine Mammal Science 22: 446–464.

Barlow, J. 2010. Cetacean abundance in the California Current from a 2008 ship-based line-transect survey. NOAA Technical Memorandum, NMFS, NOAA-TM-NMFS-SWFSC-456. 19 p.

Barlow, J. and R. Gisiner. 2006. Mitigating, monitoring, and assessing the effects of anthropogenic sound on beaked whales. Journal of Cetacean Research and Management 7:239 - 249.

Barlow, J. and B. L. Taylor. 2005. Estimates of sperm whale abundance in the northeastern temperate Pacific from a combined acoustic and visual survey. Marine Mammal Science 21:429-445.

Barrenas, M. L. 1994. The influence of a melanin-binding drug on temporary threshold shift in humans. Scandinavian audiology 23:93-98.

Barrett-Lennard, L. G., T. G. Smith, and G. M. Ellis. 1996. A cetacean biopsy system using lightweight pneumatic darts, and its effect on the behavior of killer whales. Marine Mammal Science 12:14.

Bartol, S.M. and D.R. Ketten. 2006. Turtle and tuna hearing. In: Swimmer, Y. and R. Brill, eds. Sea turtle and pelagic fish sensory biology: Developing techniques to reduce sea turtle bycatch in longline fisheries. NOAA Tech. Mem. NMFS-PIFSC-7. Pp. 98-105. Available at: http://www.pifsc.noaa.gov/tech/NOAA_Tech_Memo_PIFSC_7.pdf.

Bartol, S. M., Musick, J. A. and Lenhardt, M. L. (1999). Auditory evoked potentials of the loggerhead sea turtle (Caretta caretta). Copeia. 1999(3), 836-840.

Bass, A.L., S.P. Epperly, and J. Braun-McNeill. 2004. Multiyear analysis of stock composition of a loggerhead turtle (Caretta caretta) foraging habitat using maximum likelihood and Bayesian methods. Conservation Genetics 5:783-796.

Bearhop, S., G. M. Hilton, S. C. Votier, and S. Waldron. 2004. Stable isotope ratios indicate that body condition in migrating passerines is influenced by winter habitat. Proceedings of the Royal Society Biological Sciences Series B 271, Suppl. 4:S215-S218.

Beedholm, K. and B. Mo-hl. 2006. Directionality of sperm whale sonar clicks and its relation to piston radiation theory. The Journal of the Acoustical Society of America 119:EL14.

Benoit-Bird, K., W. Au, C. Kelley, and C. Taylor. 2002. Measuring acoustic backscattering of deepwater fish in situ using a manned submersible. The Journal of the Acoustical Society of America 112:2207.

Bentivegna, F., G. Treglia, and S. Hochscheid. 2005. The first report of a loggerhead turtle Caretta caretta nest on the central Tyrrhenian coast (western Mediterranean). Journal of the Marine Biological Association of the United Kingdom - Biodiversity Records 85(6):1555.

Berrow, S. D., P. G. H. Evans, and M. L. Sheldrick. 1993. An analysis of sperm whale *Physeter macrocephalus* stranding and sighting records, from Britain and Ireland. Journal of Zoology 230:333-337.

Best, P. B., R. A. Rademeyer, C. Burton, D. Ljungblad, S. Keiko, S. Hiroyuki, D. Thiele, D. Reeb, and D. S. Butterworth. 2003. The abundance of blue whales on the Madagascar Plateau, December 1996. Journal of Cetacean Research and Management 5:253-260.

Best, P. B., D. Reeh, M. B. Rew, P. J. Palsboll, C. Schaeff, and A. Brandao. 2005. Biopsying southern right whales: their reactions and effects on reproduction. The Journal of Wildlife Management 69:1171 - 1180.

Best, P. B. and H. Ruther. 1992. Aerial photogrammetry of southern right whales, *Eubalaena australis*. Journal of Zoology 228:595.

Biassoni, N., P. J. O. Miller, and P. L. Tyack. 2001. Humpback whales, *Megaptera novaeangliae*, alter their song to compensate for man-made noise.*in* 14th Biennial Conference on the Biology of Marine Mammals, Vancouver, Canada.

Bjerager, P., S. Heegaard, and J. Tougaard. 2003. Anatomy of the eye of the sperm whale (*Physeter macrocephalus* L). Aquatic Mammals 29:31-36.

Bjorndal, K. A., A. B. Bolten and Southeast Fisheries Science Center (U.S.) (2000). Proceedings of a workshop on Assessing Abundance and Trends for In-Water Sea Turtle Populations : held at the Archie Carr Center for Sea Turtle Research University of Florida, Gainesville, Florida, 24-26 March 2000. Miami, Fla., U.S. Department of commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center.

Blackmer, A. L., S. K. Anderson, and M. T. Weinrich. 2000. Temporal variability in features used to photo-identify humpback whales (*Megaptera novaeangliae*). Marine Mammal Science 16:17.

Bland, K. P. and A. C. Kitchener. 2001. The anatomy of the penis of a sperm whale (*Physeter catodon* L., 1758). Mammal Review 31:239-244.

Blees, M. K., K. G. Hartin, D. S. Ireland, and D. Hannay, editors. 2010. Marine mammal monitoring and mitigation during open water seismic exploration by Statoil USA E&P Inc. in the Chukchi Sea, August–October 2010: 90-day report. Report P1119 prepared by LGL Alaska Research Associates Inc., LGL Ltd., and JASCO Research Ltd. for Statoil USA E&P Inc., U.S. National Marine Fisheries Service, and U.S. Fish and Wildlife Service.

Bloch, D., G. Desportes, A. Petersen, and J. Sigurjonsson. 1996. Strandings of striped dolphins (*Stenella coeruleoalba*) in Iceland and the Faroe Islands and sightings in the northeast Atlantic, north of 50°N. Marine Mammal Science 12:8.

Blumenthal, J.M., J.L. Solomon, C.D. Bell, T.J. Austin, G. Ebanks-Petrie, M.S. Coyne, A.C. Broderick, and B.J. Godley. (2006). Satellite tracking highlights the need for international cooperation in marine turtle management. Endangered Species Research 2:51-61.

Boer, M. 2000. A note on cetacean observations in the Indian Ocean Sanctuary and the South China Sea, Mauritius to the Philippines, April 1999. Journal of Cetacean Research and Management 2.

Bolten, A. B., K. A. Bjorndal and H. R. Martins (1994). Life history model for the loggerhead sea turtle (Caretta caretta) populations in the Atlantic: Potential impacts of a longline fishery. NOAA Technical Memo, U.S. Department of Commerce.

Bonte, D., L. Lens, J.-P. Maelfait, M. Hoffmann, and E. Kuijken. 2003. Patch quality and connectivity influence spatial dynamics in a dune wolfspider. Oecologia 135:227-233.

Boon, J. P., W. E. Lewis, and A. Goksoyr. 2001. Immunochemical and catalytic characterization of hepatic microsomal cytochrome P450 in the sperm whale (*Physeter macrocephalus*). Aquatic Toxicology 52:14.

Bor, P. H. F. and M. B. Santos. 2003. Findings of elasmobranch eggs in the stomachs of sperm whales and other marine organisms. Journal of the Marine Biological Association of the UK 83:3.

Boran, J. R. and S. L. Heimlich. 1999. Part 5: Dolphins and whales: Communication and foraging in aquatic environments - 16 Social learning in cetaceans: Hunting, hearing and hierarchies. Symposia of the Zoological Society of London:26.

Borsani, J. F. and C. W. Clark. 2003. Does shipping noise in the Ligurian sea compete with fin whale song?*in* 17th Conference of the European Cetacean Society. European Cetacean Society, Las Palmas de Gran Canaria, Canary Islands, Spain.

Bowen, B.W., and S.A. Karl. (2007). Population genetics and phylogeography of sea turtles. Molecular Ecology 16:4886-4907.

Bowen, B.W., F.A. Abreu-Grobois, G.H. Balazs, N. Kamezaki, C.J. Limpus, and R.J. Ferl. (1995). Trans-Pacific migrations of the loggerhead turtles (Caretta caretta) demonstrated with mitochondrial DNA markers. Proceedings of the National Academy of Sciences of the United States of America 92:3731-3734.

Bowen, B.W., A.L. Bass, S.-M. Chow, M. Bostrom, K.A. Bjorndal, A.B. Bolten, T. Okuyama, B.M. Bolker., S. Epperly, E. Lacasella, D. Shaver, M. Dodd, S.R. Hopkins-Murphy, J.A. Musick, M. Swingle, K. Rankin-Baransky, W. Teas, W.N. Witzell, and P.H. Dutton. (2004). Natal homing in juvenile loggerhead turtles (Caretta caretta). Molecular Ecology 13:3797- 3808.

Bowles, A. E., M. Smultea, B. Wuersig, and D. P. DeMaster. 1994. Relative abundance and behavior of marine mammals exposed to transmissions from the Heard Island Feasibility Test. The Journal of the Acoustical Society of America 96:2469.

Bowles, D. 1999. An overview of the concentrations and effects of metals in cetacean species. Journal of Cetacean Research and Management Supplement Part 1:125 - 148.

Boyle, M.C., N.N. FitzSimmons, C.J. Limpus, S. Kelez, X. Velez-Zuazo, and M. Waycott. (2009). Evidence for transoceanic migrations by loggerhead sea turtles in the southern Pacific Ocean. Proceedings of the Royal Society B 276:1993-1999.

Brabyn, M. W. and I. G. McLean. 1992. Oceanography and coastal topography of herdstranding sites for whales in New Zealand. Journal of Mammalogy 73:469-476.

Branch, T. A. and D. S. Butterworth. 2001. Estimates of abundance south of 60°S for cetacean species sighted frequently on the 1978/79 to 1997/98 IWC/IDCR-SOWER sighting surveys. Journal of Cetacean Research and Management 3:251 - 270.

Branch, T. A. and D. S. Butterworth. 2001. Southern Hemisphere minke whales: standardised abundance estimates from the 1978/79 to 1997/98 IDCR-SOWER surveys. Journal of Cetacean Research and Management 3.

Braun-McNeill, J., and S.P. Epperly. (2004). Spatial and temporal distribution of sea turtles in the western North Atlantic and the U.S. Gulf of Mexico from Marine Recreational Fishery Statistics Survey (MRFSS). Marine Fisheries Review 64(4):50-56.

Braun-McNeill J., S.P. Epperly, L. Avens, M.L. Snover, and J.C. Taylor. (2008). Life stage duration and variation in growth rates of loggerhead (Caretta caretta) sea turtles from the western North Atlantic. Herpetological Conservation and Biology 3(2):273-281.

Brennin, R., B. W. Murray, M. K. Friesen, L. D. Maiers, J. W. Clayton, and B. N. White. 1997. Population genetic structure of beluga whales (Delphinapterus leucas): Mitochondrial DNA sequence variation within and among North American populations. The Canadian Journal of Zoology 75:8.

Brian, C. (2007). Loggerhead nesting in Angola: personal communication. Page 42 in State of the World's Sea Turtles: Volume II. Conservation International, Crystal City, Virginia.

Briggs, J.C. (1974). Marine Zoogeography. McGraw-Hill, New York. 475 pages.

Brill, R. L., J. L. Pawloski, D. A. Helweg, W. W. Au, and P. W. B. Moore. 1992. Target detection, shape discrimination, and signal characteristics of an echolocating false killer whale (*Pseudorca crassidens*). The Journal of the Acoustical Society of America 92:1324.

Brodie, P. F. 1993. Noise generated by the jaw actions of feeding fin whales. The Canadian Journal of Zoology 71:2546-2550.

Brown, M. W., R. Helbig, P. T. Boag, D. Gaskin, and B. N. White. 1991. Sexing beluga whales (*Delphinapterus leucas*) by means of DNA markers. The Canadian Journal of Zoology 69:1971-1976.

Brown, J.J. and G. M. Murphy. 2007. Atlantic sturgeon vessel-strike mortalities in the Delaware Estuary. Fisheries, Vol 35:2, 72-83.

Brownell, R. L., Jr., P. J. Clapham, T. Miyashita, and T. Kasuya. 2001. Conservation status of North Pacific right whales. Journal of Cetacean Research and Management Supplement 2:269 - 286.

Brownell, R. L., T. Yamada, J. G. Mead, and A. L. van Helden. 2004. Mass strandings of Cuvier's beaked whales in Japan: US Naval acoustic link? Unpublished paper SC/56/E37 presented to the International Whaling Commission's Scientific Committee, July 2004, International Whaling Commission, Cambridge, United Kingdom.

Bryant, P. J. 1995. Dating remains of gray whales from the eastern North Atlantic. Journal of Mammalogy 76:5.

Buck, J. R. and P. L. Tyack. 2000. Response of gray whales to low-frequency sounds. The Journal of the Acoustical Society of America 107:2774.

Buckland, S. T. and J. M. Breiwick. 2002. Estimated trends in abundance of eastern Pacific gray whales from shore counts (1967/68 to 1995/96). Journal of Cetacean Research and Management 4.

Bulger, A. J., B. J. Cosby, and J. R. Webb. 2000. Current, reconstructed past, and projected future status of brook trout (*Salvelinus fontinalis*) streams in Virginia. Canadian Journal of Fisheries and Aquatic Sciences 57:1515-1523.

Burdett, L. G. and W. E. McFee. 2004. Bycatch of bottlenose dolphins in South Carolina, USA, and an evaluation of the atlantic blue crab fishery categorisation. Journal of Cetacean Research and Management 6:231 - 240.

Butterworth, D. S., J. L. Korrubel, and A. E. Punt. 2002. What is needed to make a simple density-dependent response population model consistent with data for eastern North Pacific gray whales? Journal of Cetacean Research and Management 4.

Bushnoe T. M., J. A. Musick, D. S. Ha. 2005. Essential spawning and nursery habitat of Atlantic sturgeon (Acipenser oxyrinchus oxyrinchus) in Virginia. VIMS special Scientific Report #145. 44pp.

Calambokidis, J., T. Chandler, and A. E. Douglas. 2002. Marine mammal observations and mitigation associated with USGS seismic-reflection surveys in the Santa Barbara Channel 2002. Unpublished report prepared for the U.S. Department of the Interior, Geological Survey, Menlo Park, California, Cascadia Research, Olympia, Washington.

Calambokidis, J., T. E. Chandler, D. P. Costa, C. W. Clark, and H. Whitehead. 1998. Effects of ATOC sound source on the distribution of marine mammals observed from aerial surveys off central California.*in* World Marine Mammal Science Conference. Society for Marine Mammalogy, Monaco.

Calambokidis, J., E. Falcone, A. Douglas, L. Schlender, and J. Huggins. 2010. Photographic identification of humpback and blue whales off the U.S. West Coast: results and updated abundance estimates from 2008 field season. National Marine Fisheries Service Southwest Fisheries Science Center, La Jolla, CA.

Calambokidis, J., G. H. Steiger, J. R. Evenson, K. R. Flynn, K. C. Balcomb, D. E. Claridge, P. Bloedel, J. M. Straley, C. S. Baker, O. v. Ziegesar, M. E. Dahlheim, J. M. Waite, J. D. Darling, G. Ellis, and G. A. Green. 1996. Interchange and isolation of humpback whales off California and other North Pacific feeding grounds. Marine Mammal Science 12:12.

Calford, M. B., R. Rajan, and D. R. F. Irvine. 1993. Rapid changes in the frequency tuning of neurons in cat auditory cortex resulting from pure-tone-induced temporary threshold shift. Neuroscience 55:953.

Calvez, L. 2000. Deafness in the depths. Ecologist 30:58-59.

Caron, F., D. Hatin, and R. Fortin. 2002. Biological characteristics of adult Atlantic sturgeon (Acipenser oxyrinchus) in the Saint Lawrence River estuary and the effectiveness of management rules. Journal of Applied Ichthyology. 18:580-585.

Carpenter, B. 2002. Sound and fury. Pages 50-52 U S News & World Report.

Carr, S. M., H. D. Marshall, K. A. Johnstone, L. M. Pynn, and G. B. Stenson. 2002. How to tell a sea monster: Molecular discrimination of large marine animals of the North Atlantic. The Biological Bulletin 202:1-5.

Carretta, J.V., K.A. Forney, E. Oleson, K. Martien, M.M. Muto, M.S. Lowry, J. Barlow, J. Baker, B. Hanson, D. Lynch, L. Carswell, R.L. Brownell Jr., J. Robbins, D.K. Mattila, K. Ralls, and Marie C. Hill. 2012. U.S. Pacific Marine Mammal Stock Assessments: 2011. U.S. Department of Commerce, NOAA Technical Memorandum, NMFS-SWFSC-488, 356 p.

Carrier, D. R., S. M. Deban, and J. Otterstrom. 2002. The face that sank the Essex: potential function of the spermaceti organ in aggression. The journal of Experimental Biology 205:1755-1763.

Cassens, I., S. Vicario, V. G. Waddell, H. Balchowsky, D. Van Belle, W. Ding, C. Fan, R. S. Lal Mohan, P. C. Simoes-Lopes, R. Bastida, A. Meyer, M. J. Stanhope, and M. C. Milinkovitch. 2000. Independent adaptation to riverine habitats allowed survival of ancient cetacean lineages. Proceedings of the National Academy of Sciences of the United States of America 97:11343-11347.

Castellini, M. T. 2000. History of polar whaling: Insights into the physiology of the great whales. Comparative Biochemistry and Physiology Part A Molecular & Integrative Physiology 126A:153-159.

Cerchio, S., J. K. Jacobsen, D. M. Cholewiak, E. A. Falcone, and D. A. Merriwether. 2005. Paternity in humpback whales, *Megaptera novaeangliae*: assessing polygyny and skew in male reproductive success. Animal Behaviour 70:267-277.

Cerchio, S., J. K. Jacobsen, and T. F. Norris. 2001. Temporal and geographical variation in songs of humpback whales, *Megaptera novaeangliae*: Synchronous change in Hawaiian and Mexican breeding assemblages. Animal Behaviour 62:313-329.

Cetacean and Turtle Assessment Program (CeTAP). 1982. A characterization of marine mammals and turtles in the mid- and north Atlantic areas of the U.S. Outer Continental Shelf. Report prepared by the University of Rhode Island School of Oceanography for the U.S. Department of the Interior, Bureau of Land Management; Washington, D.C.

CGGVeritas. 2011. Typical seismic vessel specifications. Available at: http://oilandgastraining.org/data/gp21/P0728.asp?Code=6043.

Chambers, M. 2010. Atlantic Coast U.S. Seaports. Research and Innovative Technology Administration, Bureau of Transportation Statistics. http://www.bts.gov/publications/bts_fact_sheets/2010_002/pdf/entire.pdf

Childerhouse, S. J. and S. M. Dawson. 1996. Stability of fluke marks used in individual photoidentification of male sperm whales at Kaikoura, New Zealand. Marine Mammal Science 12:5.

Childerhouse, S. J., S. M. Dawson, and E. Slooten. 1995. Abundance and seasonal residence of sperm whales at Kaikoura, New Zealand. The Canadian Journal of Zoology 73:9.

Ching, P. 1994. The Hawaiian monk seal. University of Hawaii Press, Honolulu.

Christal, J. and H. Whitehead. 1997. Aggregations of mature male sperm whales on the Galapagos Islands breeding ground. Marine Mammal Science 13:11.

Christal, J. and H. Whitehead. 2001. Social affiliations within sperm whale (*Physeter macrocephalus*) groups. Ethology 107:18.

Christal, J., H. Whitehead, and E. Lettevall. 1998. Sperm whale social units: variation and change. The Canadian Journal of Zoology 76:1431-1440.

Christensen, I. 1990. A note on recent strandings of sperm whales (*Physeter macrocephalus*) and other cetaceans in Norwegian waters. Report of the International Whaling Commission 40:513-515.

Christian, J. R. and R. C. Bocking. 2010. Appendix D: Review of the effects of air gun sounds on marine invertebrates and fish. Programmatic EIS/OEIS NSF-Funded Marine Seismic Research.

Chu, D., A. Lavery, and J. M. Jech. 2001. Inference of size and orientation of swim-bladdered fish from high-frequency acoustic echograms. The Journal of the Acoustical Society of America 109:2336.

Ciano, J. N. and R. Huele. 2001. Photo-identification of sperm whales at Bleik Canyon, Norway. Marine Mammal Science 17:5.

Chapman, C. J. and A. D. Hawkins 1969. The importance of sound in fish behaviour in relation to capture by trawls. FAO Fisheries Report 62(3):717-729.

Clapham, P. J., C. Good, S. E. Quinn, R. R. Reeves, J. E. Scarff, and R. L. Brownell, Jr. 2004. Distribution of North Pacific right whales (*Eubalaena japonica*) as shown by 19th and 20th century whaling catch and sighting records. Journal of Cetacean Research and Management 6:1 - 6.

Clapham, P. J. and I. E. Seipt. 1991. Resightings of independent fin whales, *Balaenoptera physalus*, on maternal summer ranges. Journal of Mammalogy 72:788-790.

Clapham, P. J., S. B. Young, and R. L. Brownell, Jr. 1999. Baleen whales: Conservation issues and the status of the most endangered populations. Mammal Review 29:35-60.

Clark, C. W. 1999. On the subject of the potential impact of human-made noise on whales. Journal of

Cetacean Research and Management 1:207 - 209.

Clarke, R. 1956. Sperm whales of the Azores. Discovery Reports 28, 237-298.

Clark, C. W. and N. S. Altman. 2006. Acoustic detections of blue whale (*Balaenoptera musculus*) and fin whale (*B physalus*) sounds during a SURTASS LFA exercise. IEEE Journal of Oceanic Engineering 31:9.

Clark, C. W., J. F. Borsani, and G. Notarbartolo-di-Sciara. 2002. Vocal activity of fin whales, *Balaenoptera physalus*, in the Ligurian Sea. Marine Mammal Science 18:9.

Clark, C. W. and W. T. Ellison. 2000. Calibration and comparison of the acoustic location methods used during the spring migration of the bowhead whale, *Balaena mysticetus*, off Pt. Barrow, Alaska, 1984-1993. The Journal of the Acoustical Society of America 107:3509.

Clarke, M. R. 2003. Production and control of sound by the small sperm whales, Kogia breviceps and K sima and their implications for other Cetacea. Journal of the Marine Biological Association of the UK 83:23.

Clay, C. S. and J. K. Horne. 1994. Acoustic models of fish: The Atlantic cod (*Gadus morhua*). The Journal of the Acoustical Society of America 96:1661.

Coakes, A., S. Gowans, P. Simard, J. Giard, C. Vashro, and R. Sears. 2005. Photographic identification of fin whales (Balaenoptera physalus) off the Atlantic coast of Nova Scotia, Canada. Marine Mammal Science 21:4.

Coakes, A. and H. Whitehead. 2004. Social structure and mating system of sperm whales off northern Chile. The Canadian Journal of Zoology 82:10.

Cockcroft, V. G. 1999. Organochlorine levels in cetaceans from South Africa: a review. Journal of Cetacean Research and Management Supplement Part 1:169 - 176.

Colegrove, K. M., D. J. Greig, and F. M. D. Gulland. 2005. Causes of live strandings of northern elephant seals (*Mirounga angustirostris*) and Pacific harbor seals (*Phoca vitulina*) along the central California coast, 1992-2001. Aquatic Mammals 31:1-10.

Collins, M.R., S.G. Rogers, T.I.J. Smith, and M.L. Moser. 2000. Primary factors affecting sturgeon populations in the southeastern United States: fishing mortality and degradation of essential habitats. Bulletin of Marine Science 66(3):917-928.

Connor, R. C. 2000. Group living in whales and dolphins. Pages 199-218 *in* J. Mann, R. C. Connor, P. L. Tyack, and H. Whitehead, editors. Cetacean societies. Field studies of dolphins and whales. University of Chicago Press, Chicago, Illinois.

Connor, R. C., J. Mann, P. L. Tyack, and H. Whitehead. 2000. Introduction: the social lives of whales and dolphins. Pages 1-6 *in* J. Mann, R. C. Connor, P. L. Tyack, and H. Whitehead, editors. Cetacean societies. Field studies of dolphins and whales. University of Chicago Press, Chicago, Illinois.

Connor, R. C., A. n. J. Read, and R. Wrangham. 2000. Male reproductive strategies and social bonds. Pages 247-269 *in* J. Mann, R. C. Connor, P. L. Tyack, and H. Whitehead, editors. Cetacean societies. Field studies of dolphins and whales. University of Chicago Press, Chicago, Illinois.

Connor, R. C., R. S. Wells, J. Mann, and A. n. J. Read. 2000. The bottlenose dolphin: social relationships in a fission-fusion society. Pages 91-126 *in* J. Mann, R. C. Connor, P. L. Tyack, and H. Whitehead, editors. Cetacean societies. Field studies of dolphins and whales. University of Chicago Press, Chicago, Illinois.

Constantine, R. and D. Brunton. 2001. Boats and bottlenose dolphin (*Tursiops truncatus*) in the Bay of Islands, New Zealand. Page 46 *in* 14th Biennial Conference on the Biology of Marine Mammals, Vancouver, Canada.

Continental Shelf Associates, Inc. 2006. Effects of oil and gas exploration and development at selected continental slope sites in the Gulf of Mexico. Volume II: Technical report. U.S. Dept. of the

Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, LA. OCS Study MMS 2006-045. 636 pp. Available at: http://www.gomr.mms.gov/PI/PDFImages/ESPIS/3/3875.pdf.

Conversi, A., S. Piontkovski, and S. Hameed. 2001. Seasonal and interannual dynamics of Calanus finmarchicus in the Gulf of Maine (Northeastern US shelf) with reference to the North Atlantic Oscillation. Deep Sea Research Part Ii: Topical studies in Oceanography 48(1-3)519-530.

Corkeron, P. J. 1995. Humpback whales (*Megaptera novaeangliae*) in Hervey Bay, Queensland - behavior and responses to whale-watching vessels. The Canadian Journal of Zoology 73:1290-1299.

Corkeron, P. J. and R. C. Connor. 1999. Why do baleen whales migrate? Marine Mammal Science 15:18.

Cox, T. M., T. J. Ragen, A. J. Read, E. Vos, R. W. Baird, K. C. Balcomb, J. Barlow, J. Caldwell, T. Cranford, L. Crum, A. D'Amico, G. D. D'Spain, A. Fernandez, J. J. Finneran, R. L. Gentry, W. Gerth, F. M. D. Gulland, J. Hildebrand, D. Houser, T. Hullar, P. D. Jepson, D. R. Ketten, C. D. MacLeod, P. Miller, S. Moore, D. C. Mountain, D. Palka, P. Ponganis, S. Rommel, T. Rowles, P. L. Tyack, D. Wartzok, R. Gisiner, J. Mead, and L. Benner. 2006. Understanding the impacts of anthropogenic sound on beaked whales. Journal of Cetacean Research and Management 7:177 - 187.

Cox, T. M., A. J. Read, A. Solow, and N. Trengenza. 2001. Will harbour porpoises (*Phocoena phocoena*) habituate to pingers? Journal of Cetacean Research and Management 3.

Craig, A. S., L. M. Herman, and A. A. Pack. 2002. Male mate choice and male-male competition coexist in the humpback whale (*Megaptera novaeangliae*). The Canadian Journal of Zoology 80:745-755.

Cranford, T. W. 1999. The sperm whale's nose: Sexual selection on a grand scale? Marine Mammal Science 15:25.

Croll, D. A., A. Acevedo-Gutierrez, B. R. Tershy, and J. Urban-Ramirez. 2001. The diving behavior of blue and fin whales: Is dive duration shorter than expected based on oxygen stores? Comparative Biochemistry and Physiology Part A Molecular & Integrative Physiology 129A:797-809.

Croll, D. A., C. W. Clark, A. Acevedo, B. Tershy, S. Flores, J. Gedamke, and J. Urban. 2002. Only male fin whales sing loud songs. Page 809 Nature (London).

Croll, D. A., C. W. Clark, J. Calambokidis, W. T. Ellison, and B. R. Tershy. 2001. Effect of anthropogenic low-frequency noise on the foraging ecology of *Balaenoptera* whales. Animal Conservation 4:15.

Crouse, D. T. (1999). "Population modeling implications for Caribbean hawksbill sea turtle management. " Chelonian Conservation and Biology **3**(2): 185-188.

Crum, L. A., M. R. Bailey, J. Guan, P. R. Hilmo, S. G. Kargl, Matula, T.J., and O. A. Sapazhnikov. 2005. Monitoring bubble growth in supersaturated blood and tissue ex vivo and the relevance to marine mammal bioeffects. Acoustic Research Letters Online 6:214-220.

Crum, L. A. and Y. Mao. 1996. Acoustically enhanced bubble growth at low frequencies and its implications for human diver and marine mammal safety. The Journal of the Acoustical Society of America 99:2898-2907.

Cumming, G. 2012. Understanding the new statistics: Effect sizes, confidence intervals, and meta-analysis. New York: Routledge. Taylor & Francis Group, 711 Third Avenue, New York, NY 10017.

Cummings, W. C. and P. O. Thompson. 1971. Underwater sounds from the blue whale, *Balaenoptera musculus*. The Journal of the Acoustical Society of America 50:1193-1198.

Cummings, W. C. and P. O. Thompson. 2000. First known study, identification, and characteristics of fin and blue-whale vocalizations in the Northeast Pacific with historical notes and anecdotes. The Journal of the Acoustical Society of America 108:2612.

Curry, B. E. 1999. Stress in mammals: the potential influence of fishery-induced stress on dolphins in the eastern tropical Pacific Ocean. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center, La Jolla, California.

Curtis, K. B., B. M. Howe, and J. A. Mercer. 1999. Low-frequency ambient sounds in the North Pacific long time series observations. The Journal of the Acoustical Society of America 106:3189-3200.

D’Amico, A. and W. Verboom. 1998. Summary record and report of the SACLANTCEN Bioacoustics, Marine Mammal Policy, and Mitigation Procedures Panels, 15-19 June 1998. SACLANTCEN Marine Mammal Environmental Policy and SACLANTCEN Marine Mammal and Human Divers: Risk Mitigation Rules. SACLANTCEN M-133, SACLANCT Undersea Research Center, La Spezia, Italy.

Dabelsteen, T., O. N. Larsen, and S. B. Pedersen. 1993. Habitat-induced degradation of sound signals: Quantifying the effects of communication sounds and bird location on blur ratio, excess attenuation, and signal-to-noise ratio in blackbird song. The Journal of the Acoustical Society of America:2206.

D'Amico, A., A. Bergamasco, P. Zanasca, S. Carniel, E. Nacini, N. Portunato, V. Teloni, C. Mori, and R. Barbanti. 2003. Qualitative correlation of marine mammals with physical and biological parameters in the Ligurian Sea. IEEE Journal of Oceanic Engineering 28:29 - 43.

Dancer, A., P. Grateau, A. Cabanis, T. Vaillant, and D. Lafont. 1991. Delayed temporary threshold shift induced by impulse noises (weapon noises) in men. Audiology 30:345-356.

De Rosny, J. and P. Roux. 2001. Multiple scattering in a reflecting cavity: Application to fish counting in a tank. The Journal of the Acoustical Society of America 109:11.

Debrot, A. O. and N. B. Barros. 1992. Notes on a Gervais' beaked whale, *Mesoplodon europaeus*, and a dwarf sperm whale, *Kogia simus*, stranded in Curaco, Netherlands Antilles. Marine Mammal Science 8:172-177.

Delaugerre, M. and C. Cesarini. (2004). Confirmed nesting of the loggerhead turtle in Corsica. Marine Turtle Newsletter 104:12.

Dalen, J. and G.M. Knutsen. 1986. Scaring effects in fish and harmful effects on eggs, larvae and fry by offshore seismic explorations. Symposium on Underwater Acoustics, Halifax.

Dalen, J. and A. Raknes. 1985. Scaring effects on fish from three dimensional seismic surveys. Institute of Marine Research Report FO 8504/8505, Bergen, Norway.

Deecke, V. B., J. K. B. Ford, and P. J. B. Slater. 2005. The vocal behaviour of mammal-eating killer whales: communicating with costly calls. Animal Behaviour 69:395-405.

Deecke, V. B., J. K. B. Ford, and P. Spong. 1999. Quantifying complex patterns of bioacoustic variation: Use of a neural network to compare killer whale (*Orcinus orca*) dialects. The Journal of the Acoustical Society of America 105:2499.

Deecke, V. B., J. K. B. Ford, and P. Spong. 2000. Dialect change in resident killer whales: Implications for vocal learning and cultural transmission. Animal Behaviour 60:629-638.

Demmer, B. 2001. Time to turn off the LFA sonar. The Humanist 61:2.

DeNardo, C., M. Dougherty, G. Hastie, R. Leaper, B. Wilson, and P. M. Thompson. 2001. A new technique to measure spatial relationships within groups of free-ranging coastal cetaceans. The Journal of Animal Ecology 38:888-895.

DeRuiter, S.L and K.L. Doukara. 2012. Loggerhead turtles dive in response to airgun sound exposure. Endangered Species Research Vol. 16:55-63.

Dew, L. A., R. G. Owen, Jr., and M. J. Mulroy. 1993. Changes in size and shape of auditory hair cells in vivo noise-induced temporary threshold shift. Hearing research 66:99.

DFO 2004. Review of scientific information on impacts of seismic sound on fish, invertebrates, marine turtles and marine mammals. Department of Fisheries and Oceans, Canada. Habitat Status Report 2004/002. 15p.

Diachok, O. 1999. Effects of absorptivity due to fish on transmission loss in shallow water. The Journal of the Acoustical Society of America 105:22.

Diachok, O. 2000. Comparison of measured and calculated absorption spectra due to ensembles of fish with swim bladders. The Journal of the Acoustical Society of America 108:1.

Diachok, O. 2002. Sound absorption due to fish: From David Weston's discoveries to recent developments. The Journal of the Acoustical Society of America 112:2206.

Diachok, O. 2003. Acoustic method and system for measuring fish population data in littoral environments. The Journal of the Acoustical Society of America 114:23.

Ding, L. 1997. Direct laboratory measurement of forward scattering by individual fish. The Journal of the Acoustical Society of America 101:7.

Ding, L., Y. Takao, K. Sawada, T. Okumura, Y. Miyanohana, M. Furusawa, and D. M. Farmer. 1998. Laboratory measurements of forward and bistatic scattering of fish at multiple frequencies. The Journal of the Acoustical Society of America 103:4.

Dodd, C. K. (1988). Synopsis of the biological data on the loggerhead sea turtle: Caretta caretta (Linnaeus, 1758). Washington, D.C., Fish and Wildlife Service, U.S. Dept. of the Interior.

Domenici, P., R. S. Batty, T. Simila, and E. Ogam. 2000. Killer whales (*Orcinus orca*) feeding on schooling herring (*Clupea harengus*) using underwater tail-slaps: Kinematic analyses of field observations. The journal of Experimental Biology 203:283-294.

Donoso, M. and P.H. Dutton. (2006). Distribution and stock origin of sea turtles caught incidentally in the Chilean longline fishery for swordfish, 2001-2004. Pages 242-243 in Frick, M., A. Panagopoulou, A.F. Rees, and K. Williams (compilers). Book of Abstracts of the Twenty-sixth Annual Symposium on Sea Turtle Biology and Conservation. International Sea Turtle Society, Athens, Greece.

Donoso, M. and P.H. Dutton. (2007). Numbers, distribution and stock origin of sea turtles caught incidentally in the Chilean longline fishery for swordfish, 2001-2002. Pages 15-16 in Mast, R.B., B.J. Hutchinson, and A.H. Hutchinson (compilers). Proceedings of the Twenty-fourth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-567.

Dos Santos, R. A. and M. Haimovici. 2002. Cephalopods in the trophic relations off southern Brazil. Bulletin of Marine Science 71:753-770.

Dovel, W. L., and T. J. Berggren. 1983. Atlantic sturgeon of the Hudson River estuary, New York. New York Fish and Game Journal 30:140-172.

Drinkwater, K. F., and coauthors. 2003. The response of marine ecosystems to climate variability associated with the North Atlantic oscillation. Geophysical Monograph 134:211-234.

Drouot, V., M. Berube, A. Gannier, J. C. Goold, R. J. Reid, and P. J. Palsboll. 2004. A note on genetic isolation of Mediterranean sperm whales (Physeter macrocephalus) suggested by mitochondrial DNA. Journal of Cetacean Research and Management 6:29 - 32.

Drouot, V., A. Gannier, and J. C. Goold. 2004. Diving and feeding behaviour of sperm whales (*Physeter macrocephalus*) in the northwestern Mediterranean Sea. Aquatic Mammals 30:419-426.

Drouot, V., A. Gannier, and J. C. Goold. 2004. Summer social distribution of sperm whales (*Physeter macrocephalus*) in the Mediterranean Sea. Journal of the Marine Biological Association of the UK 84:6.

Drouot, V., J. C. Goold, and A. Gannier. 2004. Regional diversity in the social vocalizations of sperm whale in the Mediterranean Sea. Revue d'âecologie : la terre et la vie 59:545-558.

D'Spain, G. D., A. D'Amico, and D. M. Fromm. 2006. Properties of the underwater sound fields during some well documented beaked whale mass stranding events. Journal of Cetacean Research and Management 7:223 - 238.

Dufault, S. and H. Whitehead. 1995. An assessment of changes with time in the marking patterns used for photoidentification of individual sperm whales, *Physeter macrocephalus*. Marine Mammal Science 11:9.

Dufault, S. and H. Whitehead. 1995. An encounter with recently wounded sperm whales (*Physeter macrocephalus*). Marine Mammal Science 11:4.

Dufault, S. and H. Whitehead. 1998. Regional and group-level differences in fluke markings and notches of sperm whales. Journal of Mammalogy 79:514-520.

Dufault, S., H. Whitehead, and M. Dillon. 1999. An examination of the current knowledge on the stock structure of sperm whales (*Physeter macrocephalus*) worldwide. Journal of Cetacean Research and Management 1:1-10.

Duncan, A. J., R. D. McCauley, and A. L. Maggi. 2004. Predicting the environmental impact of active sonar. AIP Conference Proceedings 728:280-287.

Edds, P. L. 1987. Occurrence and general behavior of balaenopterid cetaceans summering in the St. Lawrence estuary, Canada. Canadian Journal of Zoology 65:1363 - 1376.

Eguchi, T. 2002. A method for calculating the effect of a die-off from stranding data. Marine Mammal Science 18:12.

Ehrenberg, J. E. and T. W. Steig. 2000. A method for estimating the accuracy of the three-dimensional position of acoustically tagged fish. The Journal of the Acoustical Society of America 108:2459.

Ehrhart, L.M., D.A. Bagley, and W.E. Redfoot. (2003). Loggerhead turtles in the Atlantic Ocean: geographic distribution, abundance, and population status. Pages 157-174 in A.B. Bolten and B.E. Witherington, eds. Loggerhead Sea Turtles. Washington, D.C.: Smithsonian Institution Press.

Ellis, R. 1999. Battle of the Titans: What happens when a sperm whale clashes with a giant squid in the depths of the sea? Despite fanciful stories to the contrary, the mysterious squid inevitably comes out on the losing end. National wildlife 37:28.

Ellison, W. T., K. Weixel, and C. W. Clark. 1993. Variation in received level from man-made low-frequency underwater noise sources as a function of diving animal depth. The Journal of the Acoustical Society of America 94:1850.

Elwen, S. H. and P. B. Best. 2004. Environmental factors influencing the distribution of southern right whales (Eubalaena australis) on the south coast of South Africa I: Broad scale patterns. Marine Mammal Science 20:16.

Elwen, S. H. and P. B. Best. 2004. Environmental factors influencing the distribution of southern right whales (Eubalaena australis) on the south coast of South Africa II: Within bay distribution. Marine Mammal Science 20:19.

Engås, A., S. Løkkeborg, E. Ona, and A.V. Soldal. 1996. Effects of seismic shooting on local abundance and catch rates of cod (*Gadus morhua*) and haddock (*Melanogrammus aeglefinus*). Canadian Journal of Fisheries and Aquatic Sciences 53:2238-2249.

Environmental Protection Ageny (EPA). 2003. Biological Evaluation for the Issuance of Ambient Water Quality Criteria for Dissolved Oxygen, Water Clarity and Chlorophyll a for the Chesapeake Bay and Its Tidal Tributaries U.S. Environmental Protection Office.

EPA. 2010. Future Climate Change. http://epa.gov/climatechange/science/future.html.

Encalada, S.E., K.A. Bjorndal, A.B. Bolten, J.C. Zurita, B. Schroeder, E. Possardt, C.J. Sears, and B.W. Bowen. (1998). Population structure of loggerhead turtle (Caretta caretta) nesting colonies in the Atlantic and Mediterranean as inferred from mitochondrial DNA control region sequences. Marine Biology 130:567-575.

Epperly, S.P., J. Braun, and A.J. Chester. (1995a). Aerial surveys for sea turtles in North Carolina inshore waters. Fishery Bulletin 93:254-261.

Epperly, S.P., J. Braun, A.J. Chester, F.A. Cross, J.V. Merriner, and P.A. Tester. (1995b). Winter distribution of sea turtles in the vicinity of Cape Hatteras and their interactions with the summer flounder trawl fishery. Bulletin of Marine Science 56(2):547-568.

Epperly, S. P., J. Braun, A. J. Chester, F. A. Cross, J. V. Merriner, P. A. Tester, and J. H. Churchill. 1996. Beach strandings as an indicator of at-sea mortality of sea turtles. Bulletin of Marine Science 59:289-297.

Epperly, S.P., J. Braun, and A. Veishlow. (1995c). Sea turtles in North Carolina waters. Conservation Biology 9(2):384-394.

Erbe, C. 2002. Underwater noise of whale-watching boats and potential effects on killer whales (*Orcinus orca*), based on an acoustic impact model. Marine Mammal Science 18:394-418.

Erbe, C. and D. M. Farmer. 2000. A software model to estimate zones of impact on marine mammals around anthropogenic noise. The Journal of the Acoustical Society of America 108:1327.

Erbe, C. and D. M. Farmer. 2000. Zones of impact around icebreakers affecting beluga whales in the Beaufort Sea. The Journal of the Acoustical Society of America 108:1332.

Erbe, C., A. R. King, M. Yedlin, and D. M. Farmer. 1999. Computer models for masked hearing experiments with beluga whales (*Delphinapterus leucas*). The Journal of the Acoustical Society of America 105:2967.

Evans, K., M. Hindell, and G. Hince. 2004. Concentrations of organochlorines in sperm whales (Physeter macrocephalus) from Southern Australian waters. Marine Pollution Bulletin 48:486-503.

Evans, K. and M. A. Hindell. 2004. The age structure and growth of female sperm whales (Physeter macrocephalus) in southern Australian waters. Journal of Zoology 263:14.

Evans, K., M. A. Hindell, K. Robertson, C. Lockyer, and D. Rice. 2002. Factors affecting the precision of age determination of sperm whales (*Physeter macrocephalus*). Journal of Cetacean Research and Management 4:193 - 201.

Evans, K., M. A. Hindell, and D. Thiele. 2003. Body fat and condition in sperm whales, *Physeter macrocephalus*, from southern Australian waters. Comparative Biochemistry and Physiology Part A Molecular & Integrative Physiology 134A:847-862.

Evans, K., M. Morrice, M. Hindell, and D. Thiele. 2002. Three mass strandings of sperm whales (*Physeter macrocephalus*) in southern Australian waters. Marine Mammal Science 18:22.

Evans, K. and K. Robertson. 2001. A note on the preparation of sperm whale (*Physeter macrocephalus*) teeth for age determination. Journal of Cetacean Research and Management 3:101 - 107.

Evans, P. G. H. and P. S. Hammond. 2004. Monitoring cetaceans in European waters. Mammal Review 34:131-156.

Farmer, D. M., M. V. Trevorrow, and B. Pedersen. 1999. Intermediate range fish detection with a 12-kHz sidescan sonar. The Journal of the Acoustical Society of America 106:10.

Félix, F. 2001. Observed changes of behavior in humpback whales during whalewatching encounters off Ecuador.*in* 14th Biennial Conference on the Biology of Marine Mammals, Vancouver, Canada.

Ferber, D. 2005. Sperm whales bear testimony to worldwide pollution. Science 309:1166.

Fernandez, A. 2004. Pathological findings in stranded beaked whales during the naval military manoeuvers near the Canary Islands. Pages 37-40 European Cetacean Society Newsletter.

Fernandez, A., M. Arbelo, R. Deaville, I. A. P. Patterson, P. Castro, J. R. Baker, E. Degollada, H. M. Ross, P. Herraez, A. M. Pocknell, E. Rodriguez, F. E. Howie, A. Espinosa, R. J. Reid, J. R. Jaber, V. Martin, A. A. Cunningham, and P. D. Jepson. 2004. Pathology: Whales, sonar and decompression sickness (reply). Nature 428:n.

Fernandez, A., M. Arbelo, R. Deaville, I. A. P. Patterson, P. Castro, J. R. Baker, E. Degollada, H. M. Ross, P. Herraez, A. M. Pocknell, F. Rodriguez, F. E. Howie, A. Espinosa, R. J. Reid, J. R. Jaber, V. Martin, A. A. Cunningham, and P. D. Jepson. 2004. Beaked whales, sonar and decompression sickness. Nature 428:U1 - 2.

Fernandez, A., J. F. Edwards, F. Rodriguez, A. Espinosa de los Monteros, P. Herraez, P. Castro, J. R. Jaber, V. Martin, and M. Arbelo. 2005. "Gas and fat embolic syndrome" involving a mass stranding of beaked whales (Family *Ziphiidae*) exposed to anthropogenic sonar signals. Veterinary Pathology 42:446 - 457.

Ferrero, R. C., J. Hodder, and J. Cesarone. 1994. Recent strandings of rough-toothed dolphins (*Steno bredanensis*) on the Oregon and Washington coasts. Marine Mammal Science 10:114-115.

Feuillade, C. and R. W. Nero. 1998. A viscous-elastic swimbladder model for describing enhanced-frequency resonance scattering from fish. The Journal of the Acoustical Society of America 103:11.

Feuillade, C., R. W. Nero, and R. H. Love. 1996. A low-frequency acoustic scattering model for small schools of fish. The Journal of the Acoustical Society of America 99:13.

Feuillade, C., R. W. Nero, C. H. Thompson, and R. H. Love. 2002. Three applications of David Weston's work to studies of scattering from fish. The Journal of the Acoustical Society of America 112:2206.

Finkbeiner, E.M., B.P. Wallace, J.E. Moore, R.L. Lewison, L.B. Crowder, and A.J. Read. (2011). Cumulative estimates of sea turtle bycatch and mortality in USA fisheries between 1990 and 2007. Biological Conservation 144(11):2719-2727.

Finneran, J. J. 2003. Whole-lung resonance in a bottlenose dolphin (*Tursiops truncatus*) and white whale (*Delphinapterus leucas*). The Journal of the Acoustical Society of America 114:7.

Finneran, J. J., D. A. Carder, and S. H. Ridgway. 2002. Low-frequency acoustic pressure, velocity, and intensity thresholds in a bottlenose dolphin (*Tursiops truncatus*) and white whale (*Delphinapterus leucas*). The Journal of the Acoustical Society of America 111:10.

Finneran, J. J., D. A. Carder, C. E. Schlundt, and S. H. Ridgway. 2005. Temporary threshold shift in bottlenose dolphins (*Tursiops truncatus*) exposed to mid-frequency tones. The Journal of the Acoustical Society of America 118:10.

Finneran, J. J. and M. C. Hastings. 2000. A mathematical analysis of the peripheral auditory system mechanics in the goldfish (*Carassius auratus*). The Journal of the Acoustical Society of America 108:14.

Finneran, J. J., C. W. Oliver, K. M. Schaefer, and S. H. Ridgway. 2000. Source levels and estimated yellowfin tuna (*Thunnus albacares*) detection ranges for dolphin jaw pops, breaches, and tall slaps. The Journal of the Acoustical Society of America 107:8.

Finneran, J. J., C. E. Schlundt, D. A. Carder, and S. H. Ridgway. 2002. Auditory filter shapes for the bottlenose dolphin (*Tursiops truncatus*) and the white whale (*Delphinapterus leucas*) derived with notched noise. The Journal of the Acoustical Society of America 112:7.

Finneran, J. J., C. E. Schlundt, R. Dear, D. A. Carder, and S. H. Ridgway. 2000. Masked temporary threshold shift (MTTS) in odontocetes after exposure to single underwater impulses from a seismic watergun. The Journal of the Acoustical Society of America 108:2515.

Flinn, R. D., A. W. Trites, E. J. Gregr, and R. I. Perry. 2002. Diets of fin, sei, and sperm whales in British Columbia: An analysis of commercial whaling records, 1963-1967. Marine Mammal Science 18:17.

Florida Marine Research Institute. (2003). What is red tide? Florida Marine Research Institute web page.

Foote, A. D., R. W. Osborne, and A. R. Hoelzel. 2004. Whale-call response to masking boat noise. Nature 428:1.

Foote, K. G., E. Ona, and R. Toresen. 1992. Determining the extinction cross section of aggregating fish. The Journal of the Acoustical Society of America 91:1983-1989.

Forcada, J., A. Aguilar, P. Hammond, and X. Pastor. 1996. Distribution and abundance of fin whales (*Balaenoptera physalus*) in the western Mediterranean sea during the summer. Journal of Zoology 238:23.

Forney, K.A. 2007. Preliminary estimates of cetacean abundance along the U.S. west coast and within four National Marine Sanctuaries during 2005. U.S. Department of Commerce NOAA Technical Memorandum, NOAA-TM-NMFS-SWFSC-406. 27p.

Fossi, M. C., L. Marsilli, and G. Notarbartolo di Sciara. 2003. The role of skin biopsy in the detection of exposure to endocrine disrupting chemicals in Mediterranean cetaceans. Journal of Cetacean Research and Management 5:55-60.

Fowler, G. S. 1999. Behavioral and hormonal responses of Magellanic penguins (*Spheniscus magellanicus*) to tourism and nest site visitation. Biological Conservation 90:143-149.

Francis, D. T. I. and K. G. Foote. 2001. The fish swimbladder as an acoustic waveguide. The Journal of the Acoustical Society of America 109:2359.

Frankel, A. S. and C. L. Clark. 2000. Behavioral responses of humpback whales (*Megaptera novaeangliae*) to full-scale ATOC signals. The Journal of the Acoustical Society of America 108:1930-1937.

Frankel, A. S. and C. W. Clark. 1998. Results of low-frequency m-sequence noise playbacks to humpback whales in Hawaii. The Canadian Journal of Zoology 76:521-535.

Frankel, A. S. and C. W. Clark. 2002. ATOC and other factors affecting the distribution and abundance of humpback whales (*Megaptera novaeangliae*) off the north shore of Kauai. Marine Mammal Science 18:19.

Frankel, A. S., C. W. Clark, L. M. Herman, and C. M. Gabriele. 1995. Spatial distribution, habitat utilization, and social interactions of humpback whales, *Megaptera novaeangliae*, off Hawai'i, determined using acoustic. The Canadian Journal of Zoology 73:13.

Frankel, A. S. and J. Straley. 2000. Comparison of Alaskan and Hawaiian humpback whale song at the song-unit level. The Journal of the Acoustical Society of America 108:2634.

Frantzis, A. 1998. Does acoustic testing strand whales? Nature 392:29.

Frantzis, A. 2001. Sperm whales: the animals of superlatives! Acta Astronautica 49:557-561.

Frantzis, A. 2004. The first mass stranding that was associated with the use of active sonar (Kyparissiakos Gulf, Greece, 1996).*in* P. G. H. Evans and L. A. Miller, editors. Proceedings of a Workshop on Active Sonar and Cetaceans. European Cetacean Society's 17th Annual Conference, Auditorio Alfredo Kraus. European Cetacean Society Newsletter No. 42 - Special Issue, Las Palmas, Gran Canaria.

Frantzis, A., P. Alexiadou, G. Paximadis, E. Politi, A. Gannier, and M. Corsini-Foka. 2003. Current knowledge of the cetacean fauna of the Greek Seas. Journal of Cetacean Research and Management 5: 3, 219-232. .

Frantzis, A., J. C. Goold, E. K. Skarsoulis, M. I. Taroudakis, and V. Kandia. 2002. Clicks from Cuvier's beaked whales, *Ziphius cavirostris* (L). The Journal of the Acoustical Society of America 112:4.

Frantzis, A., O. Nikolaou, J. M. Bompar, and A. Cammedda. 2004. Humpback whale (*Megaptera novaeangliae*) occurrence in the Mediterranean Sea. Journal of Cetacean Research and Management 6:25 - 28.

Frazer, L. N. and E. Mercado III. 2000. A sonar model for humpback whale song. IEEE Journal of Oceanic Engineering 25:23.

Friday, N. A. and T. D. Smith. 2003. The effect of age and sex selective harvest patterns for baleen whales. Journal of Cetacean Research and Management 5: 23-28.

Friedlaender, A. S., W. A. McLellan, and D. A. Pabst. 2001. Characterising an interaction between coastal bottlenose dolphins (*Tursiops truncatus*) and the spot gillnet fishery in southeastern North Carolina, USA. Journal of Cetacean Research and Management 3.

Fristrup, K. M. and G. R. Harbison. 2002. How do sperm whales catch squids? Marine Mammal Science 18:13.

Fristrup, K. M., L. T. Hatch, and C. W. Clark. 2003. Variation in humpback whale (*Megaptera novaeangliae*) song length in relation to low-frequency sound broadcasts. The Journal of the Acoustical Society of America 113:14.

Fritts, T.H. 1983. Turtles, birds, and mammals in the northern Gulf of Mexico and nearby Atlantic waters. FWS/OBS-82/65. Report prepared for the U.S. Department of the Interior, Fish and Wildlife Service; Washington, D.C.

Fujiwara, M. and H. Caswell. 2002. A general approach to temporary emigration in mark-recapture analysis. Ecology 83:3266-3275.

Funk, D., D. Hannay, D. S. Ireland, R. Rodrigues, and W. Koski, editors. 2008. Marine mammal monitoring and mitigation during open water seismic exploration by Statoil USA E&P Inc. in the Chukchi Sea, July - November 2007: 90-day report. Report P969-1 prepared by LGL Alaska Research Associates Inc., LGL Ltd., and JASCO Research Ltd. for Shell Offshore Inc., U.S. National Marine Fisheries Service, and U.S. Fish and Wildlife Service.

Furusawa, M., K. Ishii, and Y. Miyanohana. 1992. Attenuation of sound by schooling fish. The Journal of the Acoustical Society of America 92:987.

Gabriele, C. M., J. M. Straley, S. A. Mizroch, C. S. Baker, A. S. Craig, L. M. Herman, D. Glockner-Ferrari, M. J. Ferrari, S. Cerchio, O. von Ziegesar, J. Darling, D. McSweeney, T. J. Quinn, II, and J. K. Jacobsen. 2001. Estimating the mortality rate of humpback whale calves in the central North Pacific Ocean. The Canadian Journal of Zoology 79:589-600.

Gagnon, C. J. and C. W. Clark. 1993. The use of U.S. Navy IUSS passive sonar to monitor the movement of blue whales. Abstracts of the 10th Biennial Conference on the Biology of Marine Mammals, Galveston, Texas. November 1993.

Gales, N. J., M. L. Dalebout, and J. L. Bannister. 2002. Genetic identification and biological observation of two free-swimming beaked whales: Hector's beaked whale (*Mesoplodon hectori*, Gray, 1871), and Gray's beaked whale (*Mesoplodon grayi*, von Haast, 1876). Marine Mammal Science 18:7.

Gales, R., D. Pemberton, M. Clarke, and C. C. Lu. 1992. Stomach contents of long-finned pilot whales (*Globicephala melas*) and bottlenose dolphins (*Tursiops truncatus*) in Tasmania. Marine Mammal Science 8:405-413.

Gannon, D. P., A. J. Read, J. E. Craddock, and J. G. Mead. 1997. Stomach contents of long-finned pilot whales (*Globicephala melas*) stranded on the U S mid-Atlantic coast. Marine Mammal Science 13:14.

Gardner, S. C. and S. Chavez-Rosales. 2000. Changes in the relative abundance and distribution of gray whales (*Eschrichtius robustus*) in Magdalena Bay, Mexico during an El Nino event. Marine Mammal Science 16:11.

Gauthier, J. and R. Sears. 1999. Behavioral response of four species of balaenopterid whales to biopsy sampling. Marine Mammal Science 15:17.

Gauthier, J. M., H. Dubeau, E. Rassart, W. M. Jarman, and R. S. Wells. 1999. Biomarkers of DNA damage in marine mammals. Mutation Research 444:427-439.

Gauthier, S. and G. A. Rose. 2000. Some problems and solutions for the measurement of fish target strength: A study case with Atlantic redfish (Sebastes spp). The Journal of the Acoustical Society of America 108:2458.

Gedamke, J., D. P. Costa, and A. Dunstan. 2001. Localization and visual verification of a complex minke whale vocalization. The Journal of the Acoustical Society of America 109:3038.

Geiszinger, A., S. Khokiattiwong, W. Goessler, and K. A. Francesconi. 2002. Identification of the new arsenic-containing betaine, trimethylarsoniopropionate, in tissues of a stranded sperm whale Physeter catodon. Journal of the Marine Biological Association of the UK 82:4.

Gendron, D., S. Aguiniga, and J. D. Carriquiry. 2001. delta 15N and delta 13N in skin biopsy samples: a note on their applicability for examining the relative trophic level in three rorqual species. Journal of Cetacean Research and Management 3.

Geophysical Service, Inc. 2011a. Marine seismic survey vessel: M/V *GSI Admiral*, vessel specifications. Available at:

http://www.geophysicalservice.com/Site_Files/My_Files/Seismic%20Fleet/GSI%20Admiral%20Spec%20Sheet.pdf.

Geophysical Service, Inc. 2011b. Marine seismic survey vessel: M/V GSI Pacific, vessel specifications. Available at: http://www.geophysicalservice.com/Site_Files/My_Files/Seismic%20Fleet/GSI%20Pacific%20Spec%20Sheet,%20Updated%20May%202008.pdf.

Gerstein, E. R., L. Gerstein, S. E. Forsythe, and J. E. Blue. 1999. The underwater audiogram of the West Indian manatee (*Trichechus manatus*). The Journal of the Acoustical Society of America 105:3575.

Geraci, J.R. and D.J. St. Aubin, eds. 1990. Sea mammals and oil: Confronting the risks. San Diego, CA: Academic Press.Gibson, K. R. and H. O. Box. 1999. Part 5: Dolphins and whales: Communication and foraging in aquatic environments - Editors' comments. Symposia of the Zoological Society of London:3.

Gill, P. C. 2002. A blue whale (*Balaenoptera musculus*) feeding ground in a southern Australian coastal upwelling zone. Journal of Cetacean Research and Management 4:179 - 184.

Gisiner, R. C. 1998. Workshop on the effects of anthropogenic noise in the marine environment. U.S. Navy, Office of Naval Research, Marine Mammal Research Program, Washington, D.C.

Glass AH, Cole TVN, Garron M (2009) Mortality and serious injury determinations for baleen whale stocks along the United States eastern seaboard and adjacent Canadian Maritimes, 2003-2007. *Northeast Fisheries Science Center Reference Document 09-04*. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Science Center, Woods Hole, MA.

Gonzalez, A. F., A. Lopez, and X. Valeiras. 2000. First recorded mass stranding of short-finned pilot whales (*Globicephala macrorhynchus* Gray, 1846) in the northeastern Atlantic. Marine Mammal Science 16:6.

Goodyear, J. D. 1993. A sonic/radio tag for monitoring dive depths and underwater movements of whales. The Journal of Wildlife Management 57:503-513.

Goold, J. C. 1996. Signal processing techniques for acoustic measurement of sperm whale body lengths. The Journal of the Acoustical Society of America 100:3431.

Goold, J. C. 1999. Behavioural and acoustic observations of sperm whales in Scapa Flow, Orkney Islands. Journal of the Marine Biological Association of the UK 79:10.

Goold, J. C. and M. R. Clarke. 2000. Sound velocity in the head of the dwarf sperm whale, Kogia sima, with anatomical and functional discussion. Journal of the Marine Biological Association of the UK 80:8.

Goold, J. C. and S. E. Jones. 1995. Time and frequency domain characteristics of sperm whale clicks. The Journal of the Acoustical Society of America 98:1279.

Goold, J. C., H. Whitehead, and R. J. Reid. 2002. North Atlantic sperm whale, *Physeter macrocephalus*, strandings on the coastlines of the British Iles and eastern Canada. The Canadian Field-Naturalist 116:18.

Gordon, J., J. Gillespie, J. Potter, A. Frantzis, M.P. Simmonds, R. Swift, and D. Thompson. 2004. A review of the effects of seismic surveys on marine mammals. Marine Technology Society Journal 37(4):16-34.

Gowans, S., M. Dalebout, S. Hooker, and H. Whitehead. 2000. Reliability of photographic and molecular techniques for sexing northern bottlenose whales (Hyperoodon ampullatus). The Canadian Journal of Zoology 78:6.

Gowans, S. and L. Rendell. 1999. Head-butting in northern bottlenose whales (Hyperoodon ampullatus): A possible function for big heads? Marine Mammal Science 15:9.

Gowans, S. and H. Whitehead. 2001. Photographic identification of northern bottlenose whales (Hyperoodon ampullatus): Sources of heterogeneity from natural marks. Marine Mammal Science 17:18.

Gowans, S., H. Whitehead, J. K. Arch, and S. K. Hooker. 2000. Population size and residency patterns of northern bottlenose whales (Hyperoodon ampullatus) using the Gully, Nova Scotia. Journal of Cetacean Research and Management 2.

Gowans, S., H. Whitehead, and S. K. Hooker. 2001. Social organization in northern bottlenose whales, Hyperoodon ampullatus: Not driven by deep-water foraging? Animal Behaviour 62:369-377.

Greene, C., A. J. Pershing, R. D. Kenney, and J. W. Jossi. 2003a. Impact of climate variability on the recovery of endangered North Atlantic right whales. Oceanography 16(4):98-103.

Greene, C. H., and coauthors. 2003b. Trans-Atlantic responses of Calanus finmarchicus populations to basin-scale forcing associated with the North Atlantic Oscillation. Progress in Oceanography 58(2-4):301-312.

Greene, C. R., Jr and W. J. Richardson 1988. Characteristics of marine seismic survey sounds in the Beaufort Sea. Journal of the Acoustical Society of America 83(6):2246-2254.

Green, G. A., J. J. Brueggeman, R. A. Grotefendt, C. E. Bowlby, M. L. Bonnell, and K. C. Balcomb, III. 1992. Chapter 1. Cetacean distrubition and abundance off Oregon and Washington, 1989-1990. *In*: J. J. Brueggeman, (ed). Oregon and Washington marine mammal and seabird surveys. U.S. Deparrtment of the Interior, Minerals Management Service, Pacific Outer Continental Shelf Region, Los Angeles, California.

Gregr, E. J., L. Nichol, J. K. B. Ford, G. Ellis, and A. W. Trites. 2000. Migration and population structure of northeastern Pacific whales off coastal British Columbia: An analysis of commercial whaling records from 1908-1967. Marine Mammal Science 16:29.

Gregr, E. J. and A. W. Trites. 2001. Predictions of critical habitat for five whale species in the waters of coastal British Columbia. Canadian Journal of Fisheries and Aquatic Sciences 58:1265-1285.

Greig, A. P., E. R. Secchi, A. N. Zerbini, and L. Dalla Rosa. 2001. Stranding events of southern right whales, Eubalaena australis, in southern Brazil. Journal of Cetacean Research and Management Special Issue 2:157 - 160.

Griffin, R. B. 1999. Sperm whale distributions and community ecology associated with a warm-core ring off Georges Bank. Marine Mammal Science 15:33-51.

Guigne, J. Y. and T. J. McKeever. 2002. Fish sizing sonar system. The Journal of the Acoustical Society of America 111:6.

Guinet, C., L. G. Barrett-Lennard, and B. Loyer. 2000. Co-ordinated attack behavior and prey sharing by killer whales at Crozet Archipelago: Strategies for feeding on negatively-buoyant prey. Marine Mammal Science 16:5.

Guinet, C. and J. Bouvier. 1995. Development of intentional stranding hunting techniques in killer whale (Orcinus orca) calves at Crozet Archipelago. The Canadian Journal of Zoology 73:7.

Hanni, K. D., D. J. Long, R. E. Jones, P. Pyle, and L. E. Morgan. 1997. Sightings and strandings of Guadalupe fur seals in central and northern California, 1988-1995. Journal of Mammalogy 78:684-690.

Harris, C. M. 1998. Chapter 1. Introduction. Pages 1.1 - 1.29 *in* C. M. Harris, editor. Handbook of acoustical measurements and noise control. Acoustical Society of America, Woodbury, New York.

Harris, C. M. 1998. Chapter 2. Definitions, abbreviations, and symbols. Pages 2.1 - 2.23 *in* C. M. Harris, editor. Handbook of acoustical measurements and noise control. Acoustical Society of America, Woodbury, New York.

Harris, C. M., editor. 1998. Handbook of acoustical measurements and noise control. Acoustical Society of America, Woodbury, New York.

Harris, R. E., G. W. Miller, and W. J. Richardson. 2001. Seal responses to airgun sounds during summer seismic surveys in the Alaskan Beaufort Sea. Marine Mammal Science 17:795.

Hastie, G. D., R. J. Swift, J. C. D. Gordon, G. Slesser, and W. R. Turrell. 2003. Sperm whale distribution and seasonal density in the Faroe Shetland Channel. Journal of Cetacean Research and Management 5: 3, 247-252. .

Hastings, M. C., A. N. Popper, J. J. Finneran, and P. J. Lanford. 1996. Effects of low-frequency underwater sound on hair cells of the inner ear and lateral line of the teleost fish *Astronotus ocellatus*. The Journal of the Acoustical Society of America 99:8.

Hatch, L. T. 2004. Male genes and male song: Integrating genetic and acoustic data in defining fin whale, *Balaenoptera physalus*, management units. Dissertation Abstracts International 65:277.

Hazel, J., I.R. Lawler, H. Marsh and S. Robson. 2007. Vessel speed increases collision risk for the green sea turtle *Chelonia mydas*. Endangered Species Research 3:105-113.

Hecht, J. 2002. Navy accepts blame for whale deaths. Page 15 New scientist.

Hedgepeth, J. B. and G. E. Johnson. 2000. An active fish tracking split-beam sonar to study salmon smolt behavior. The Journal of the Acoustical Society of America 108:2459.

Heide-Jorgensen, M. P., L. Kleivane, N. Oien, K. L. Laidre, and M. V. Jensen. 2001. A new technique for deploying satellite transmitters on baleen whales: Tracking a blue whale (*Balaenoptera musculus*) in the North Atlantic. Marine Mammal Science 17:949.

Hellstrom, P.-A. 1993. The relationship between sound transfer functions from free sound field to eardrum and temporary threshold shift. The Journal of the Acoustical Society of America 94:1301.

Henry AG, Cole TVN, Garron M, Hall L, Ledwell W, Reid A. 2012. Mortality and Serious Injury Determinations for Baleen Whale Stocks along the Gulf of Mexico, United States East Coast and Atlantic Canadian Provinces, 2006-2010. US Dept Commer, Northeast Fish Sci Cent Ref Doc. 12-11; 24 p. Available from: National Marine Fisheries Service, 166 Water Street, Woods Hole, MA 02543-1026, or online at http://nefsc.noaa.gov/publications/

Henshaw, M. D., R. G. LeDuc, S. J. Chivers, and A. E. Dizon. 1997. Identifying beaked whales (family Ziphiidae) using mtDNA sequences. Marine Mammal Science 13:8.

Herube, M., C. Berchok, and R. Sears. 2001. Observation of a male-biased sex ratio in the Gulf of St. Lawrence fin whales (*Balaenoptera physalus*): Temporal, geographical, or group structure segregation? Marine Mammal Science 17:371.

Heyning, J. E. and T. D. Lewis. 1990. Entanglements of baleen whales in fishing gear off southern California. Report of the International Whaling Commission 40:427-431.

Hildebrand, J. A. 2005. Annex K: Report of the standing working group on environmental concerns. Appendix 3. Introduction to acoustics. Journal of Cetacean Research and Management 7:284 - 286.

Hildebrand, J. A. 2009. Anthropogenic and natural sources of ambient noise in the ocean. Marine Ecology Progress Series 395:5-20.

Hildebrand, S.F. and W.C. Schroeder. 1928. Fishes of the Chesapeake Bay. Bull. U.S. Bureau Fish, 43 (1):1-38.

Hinkelman, L. M., T. D. Mast, L. A. Metlay, and R. C. Waag. 1998. The effect of abdominal wall morphology on ultrasonic pulse distortion. Part I. Measurements. The Journal of the Acoustical Society of America 104:3635.

Hoekstra, P. F., L. A. Dehn, J. C. George, K. R. Solomon, D. C. G. Muir, and T. M. O'Hara. 2002. Trophic ecology of bowhead whales (Balaena mysticetus) compared with that of other arctic marine biota as interpreted from carbon-, nitrogen-, and sulfur-isotope signatures. The Canadian Journal of Zoology 80:223-231.

Hoekstra, P. F., C. S. Wong, T. M. O'Hara, K. R. Solomon, S. A. Mabury, and D. C. G. Muir. 2002. Enantiomer-specific accumulation of PCB atropisomers in the bowhead whale (*Balaena mysticetus*). Environmental Science & Technology 36:1419-1425.

Hoelzel, A. R., A. Natoli, M. E. Dahlheim, C. Olavarria, R. W. Baird, and N. A. Black. 2002. Low worldwide genetic diversity in the killer whale (Orcinus orca): Implications for demographic history. Proceedings of the Royal Society Biological Sciences Series B 269:1467-1473.

Holt, S. J. 2003. The tortuous history of "scientific" Japanese whaling. Bioscience 53:204-206.

Hooker, S. K. and R. W. Baird. 1999. Deep-diving behaviour of the northern bottlenose whale, Hyperoodon ampullatus (Cetacea: Ziphiidae). Proceedings of the Royal Society Biological Sciences Series B 266:671-676.

Hooker, S. K. and R. W. Baird. 2001. Diving and ranging behaviour of odontocetes: A methodological review and critique. Mammal Review 31:81-105.

Hooker, S. K. and H. Whitehead. 2002. Click characteristics of northern bottlenose whales (*Hyperoodon ampullatus*). Marine Mammal Science 18:12.

Horne, J. and C. Clay. 1998. Sonar systems and aquatic organisms: matching equipment and model parameters. Canadian Journal of Fisheries and Aquatic Sciences 55:11.

Houde, M., L. N. Measures, and J. t. Huot. 2003. Experimental transmission of Pharurus pallasii (Nematoda: Metastrongyloidea), a lungworm of the cranial sinuses of the beluga whale (Delphinapterus leucas), to fish. The Canadian Journal of Zoology 81:364-370.

Houde, M., L. N. Measures, and J. t. Huot. 2003. Lungworm (Pharurus pallasii: Metastrongyloidea: Pseudaliidae) infection in the endangered St. Lawrence beluga whale (Delphinapterus leucas). The Canadian Journal of Zoology 81:543-551.

Houser, D. S., R. Howard, and S. Ridgway. 2001. Can diving-induced tissue nitrogen supersaturation increase the chance of acoustically driven bubble growth in marine mammals? Journal of Theoretical Biology 213:183 - 195.

Huang, C. C., T. N. Wu, and J. Y. Tung. 1994. The dose-response relationship with noise-induced temporary threshold shift (NITTS). Inter-Noise Proceedings 3:1085.

Huele, R. and H. U. D. Haes. 1998. Identification of individual sperm whales by wavelet transform of the trailing edge of the flukes. Marine Mammal Science 14:3.

Iga, K., Y. Fukui, A. Miyamoto, H. Ishikawa, and S. Ohsumi. 1996. Endocrinological observations of female minke whales (*Balaenoptera acutorostrata*). Marine Mammal Science 12:5.

Intergovernmental Panel on Climate Change (IPCC). 2007. Climate Change 2007: The synthesis report. Contribution of Working Groups I, II and III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Core Writing Team, Pachauri, R.K. and Reisinger, A. (Eds.). IPCC, Geneva, Switzerland. pp 104. Available at: www.ipcc.ch.

International Whaling Commission (IWC). 2012L. Annual Report of the International Whaling Commission 2011. Available at: www.iwcoffice.org.

IWC. 2012m. Report of the Aboriginal Subsistence Whaling Sub-Committee. IWC/64/Rep3. June 27, 2012, Panama. Available at: www.iwcoffice.org.

IWC 2012n—Proposed schedule amendment: Greenland catch limits (Denmark) IWC/64/12. May 22, 2012. Available at:http://www.iwcoffice.org/iwc64docs.

IWC 2012o. Catches taken: Under Objection. Available at: http://www.iwcoffice.org/table_objection..

IWC. 2012x. Report of the Aboriginal Subsistence Whaling Sub-Committee. IWC/64/Rep3. June 27, 2012, Panama. www.iwcoffice.org.

Ioup, J. W. and G. E. Ioup. 2004. Sperm whale identification using self-organizing maps. The Journal of the Acoustical Society of America 115:1.

Jansen, G. 1998. Chapter 25. Physiological effects of noise. Pages 25.21 - 25.19 *in* C. M. Harris, editor. Handbook of acoustical measurements and noise control. Acoustical Society of America, Woodbury, New York.

Jaquet, N., S. Dawson, and E. Slooten. 2000. Seasonal distribution and diving behaviour of male sperm whales off Kaikoura: Foraging implications. The Canadian Journal of Zoology 78:407-419.

Jaquet, N. and D. Gendron. 2002. Distribution and relative abundance of sperm whales in relation to key environmental features, squid landings and the distribution of other cetacean species in the Gulf of California, Mexico. Marine Biology 141:591-601.

Jaquet, N., D. Gendron, and A. Coakes. 2003. Sperm whales in the Gulf of California: residency, movements, behavior, and the possible influence of variation in food supply. Marine Mammal Science 19:18.

Jasny, M., J. Reynolds, C. Horowitz, and A. Wetzler. 2005. Sounding the depths II: The rising toll of sonar, shipping and industrial ocean noise on marine life. Natural Resources Defense Council, New York, New York.

Jefferson, T. A. and A. J. Schiro. 1997. Distribution of cetaceans in the offshore Gulf of Mexico. Mammal Review 27:27-50.

Jensen, H., B.-E. Saether, T. H. Ringsby, J. Tufto, S. C. Griffith, and H. Ellegren. 2004. Lifetime reproductive success in relation to morphology in the house sparrow Passer domesticus. The Journal of Animal Ecology 73:599-611.

Jepson, P. D., M. Arbelo, R. Deaville, I. A. P. Patterson, P. Castro, J. R. Baker, E. Degollada, H. M. Ross, P. Herráez, A. M. Pocknell, F. Rodriguez, F. E. Howie, A. Espinosa, R. J. Reid, J. R. Jaber, V. Martin, A. A. Cunningham, and A. Fernández. 2003. Gas-bubble lesions in stranded cetaceans. Nature 425:575-576.

Johnson, A., G. Salvador, J. Kenney, J. Robbins, S. Kraus, S. Landry, and P. Clapham. 2005. Fishing gear involved in entanglements of right and humpback whales. Marine Mammal Science 21:11.

Johnson, M., P. Tyack, D. Nowacek, and A. Shorter. 2000. A digital acoustic recording tag for measuring the response of marine mammals to sound. The Journal of the Acoustical Society of America 108:2582.

Jones, D. M. and D. E. Broadbent. 1998. Chapter 24. Human peformance and noise. Pages 24.21 - 24.24 *in* C. M. Harris, editor. Handbook of acoustical measurements and noise control. Acoustical Society of America, Woodbury, New York.

Jones, N. 2003. Is undersea noise harming whales? Page 8 New scientist.

Jørgensen, R., K. K. Olsen, I.-B. Falk-Petersen, and P. Kanapthippilai. 2005. Investigations of potential effects of low frequency sonar signals on survival, development and behavior of fish larvae and juveniles. The Norwegian College of Fishery Science, University of Tromso, Tromso, Norway.

Kahnle, A. W., K. A. Hattala, and K. McKown. 2007. Status of Atlantic sturgeon of the Hudson River estuary, New York, USA.

Kastak, D., R. J. Schusterman, B. L. Southall, and C. J. Reichmuth. 1999. Underwater temporary threshold shift induced by octave-band noise in three species of pinniped. The Journal of the Acoustical Society of America 106:1142.

Kastelein, R. A., W. W. L. Au, H. T. Rippe, and N. M. Schooneman. 1999. Target detection by an echolocating harbor porpoise (*Phocoena phocoena*). The Journal of the Acoustical Society of America 105:2493.

Kastelle, C. R., K. E. W. Shelden, and D. K. Kimura. 2003. Age determination of mysticete whales using 210Pb/226Ra disequilibria. The Canadian Journal of Zoology 81:21-32.

Kasuya, T. 1991. Density dependent growth in North Pacific sperm whales. Marine Mammal Science 7:230-257.

Kato, H., and T. Miyashita. 1998. Current status of North Pacific sperm whales and its preliminary abundance estimates. Unpubl. report submitted to Int. Whal. Comm. (SC/50/CAWS/52). 6 pp.

Kemper, C. M. and P. Leppard. 1999. Estimating body length of pygmy right whales (*Caperea marginata*) from measurements of the skeleton- and baleen. Marine Mammal Science 15:18.

Kenney, R. D. 2007. Right whales and climate change: Facing the prospect of a greenhouse future. The Urban Whale: North Atlantic Right Whales at the Crossroads. S. D. Kraus AND R. Rolland (eds.). p.436-459. Harvard University Press, Cambridge, MA. ISBN 0-674-02327-7. 543pp.

Ketten, D. R. 2005. Annex K: Report of the standing working group on environmental concerns. Appendix 4. Marine mammal auditory systems: a summary of audiometric and anatomical data and implications for underwater acoustic impacts. Journal of Cetacean Research and Management 7:286 - 289.

Kinas, P. G. 2002. The impact of incidental kills by gill nets on the Franciscana dolphin (*Pontoporia blainvillei*) in southern Brazil. Bulletin of Marine Science 70:409-421.

Klimley, A. P. and S. C. Beavers. 1998. Playback of acoustic thermometry of ocean climate (ATOC) -like signal to bony fishes to evaluate phonotaxis. The Journal of the Acoustical Society of America 104:2506.

Kloser, R. J. 2000. In situ target strength measurements of a deep-water fish, orange roughy. The Journal of the Acoustical Society of America 108:2458.

Knowlton, A. R., J. Sigurjosson, J. N. Ciano, and S. D. Kraus. 1992. Long-distance movements of North Atlantic right whales (*Eubalaena glacialis*). Marine Mammal Science 8:397-404.

Kopelman, A. H. and S. S. Sadove. 1995. Ventilatory rate differences between surface-feeding and non-surface-feeding fin whales (*Balaenoptera physalus*) in the waters off eastern Long Island, New York, U S A , 1981-1987. Marine Mammal Science 11:9.

Koski, W. R. and S. R. Johnson. 1987. Behavioral studies and aerial photogrammetry. Section 4., Report from LGL Limited and Greeneridge Science, Inc. for Shell Western Exploration and Production, Inc., Santa, Barbara, California and Anchorage, Alaska.

Kraus, S. D. 1990. Rates and potential causes of mortality in North Atlantic right whales (*Eubalaena glacialis*). Marine Mammal Science 6:278-291.

Kuczaj, S., R. Paulos, J. Ramos, R. Thames, G. Rayborn, G. Ioup, and J. Newcomb. 2003. Anthropogenic noise and sperm whale sound production.*in* 17th Conference of the European Cetacean Society, Las Palmas de Gran Canaria, Canary Islands, Spain.

Kujawa, S. G. and M. C. Liberman. 1997. Conditioning-related protection from acoustic injury: Effects of chronic deefferentation and sham surgery. Journal of Neurophysiology 78:3095-3106.

Kundzewicz, Z.W., L. J. Mata, N. W. Arnell, P. Döll, B. Jimenez, K. Miller, T. Oki, Z. Şen and I. Shiklomanov. The implications of projected climate change for freshwater resources and their management. Hydrological Sciences–Journal–des Sciences Hydrologiques, 53(1) February 2008.

Lagerquist, B. A., K. M. Stafford, and B. R. Mate. 2000. Dive characteristics of satellite-monitored blue whales (*Balaenoptera musculus*) off the central California coast. Marine Mammal Science 16:17.

Laist, D. W., A. R. Knowlton, J. G. Mead, A. S. Collet, and M. Podesta. 2001. Collisions between ships and whales. Marine Mammal Science 17:35-75.

Lambertsen, R. H. 1983. Crassicaudiasis of the North Atlantic fin whale (Balaenoptera physalus): Prevalence, pathogenesis, transmission, and life cycle. Fifth Biennial Conference on the Biology of Marine Mammals, 27 November-1 December New England Aquarium Boston MA. p.59.

Lambertsen, R. H. 1986. Disease of the common fin whale (*Balaenopters physalus*): Crassicaudiosis of the urinary system. Journal of Mammalogy 67(2):353-366.

Lambertsen, R. H. 1992. Crassicaudosis: A parasitic disease threatening the health and population recovery of large baleen whales. (*Balaenoptera musculus, Balaenoptera physalus, Megaptera novaeangliae*). Revue Scientifique Et Technique Office International Des Epizooties 11(4):1131-1141.

Lambourn, D. M., S. J. Jeffries, and J. P. Dubey. 2001. Seroprevalence of *Toxoplasma gondii* in harbor seals (*Phoca vitulina*) in Southern Puget Sound, Washington. The Journal of Parasitology 87:1196-1197.

Laplanche, C., O. Adam, M. Lopatka, and J.-F. Motsch. 2005. Male sperm whale acoustic behavior observed from multipaths at a single hydrophone. The Journal of the Acoustical Society of America 118:11.

Larue, J. P., G. E. Ioup, and J. W. Ioup. 2004. Detecting sperm whale clicks in the presence of ambient and shipping noise using higher order moments. The Journal of the Acoustical Society of America 115:1.

Lawson, G., M. Barange, and P. Freon. 2000. Species identification of pelagic fish schools using acoustic descriptors and ancillary information. The Journal of the Acoustical Society of America 108:2490.

Le Duc, R. G., W. L. Perryman, J. W. Gilpatrick Jr, J. Hyde, C. Stinchcomb, J. V. Carretta, and R. L. Brownell Jr. 2001. A note on recent surveys for right whales in the southeastern Bering Sea. Journal of Cetacean Research and Management 2:287-289.

Leaper, R., D. Gillespie, and V. Papastavrou. 2000. Results of passive acoustic surveys for odontocetes in the Southern Ocean. Journal of Cetacean Research and Management 2:187 - 196.

Leatherwood, S., R.R. Reeves, W.F. Perrin, and W.E. Evans. 1982. Whales, dolphins, and porpoises of the eastern North Pacific and adjacent arctic waters: a guide to their identification. NOAA Technical Report National Marine Fisheries Service Circular 444.

Lecchini, D., J. Shima, B. Banaigs, and R. Galzin. 2005. Larval sensory abilities and mechanisms of habitat selection of a coral reef fish during settlement. Oecologia 143:326-334.

Lees, S., D. B. Hanson, and E. A. Page. 1996. Some acoustical properties of the otic bones of a fin whale. The Journal of the Acoustical Society of America 99:2421.

Lenhardt, M. 2002. Sea turtle auditory behavior. The Journal of the Acoustical Society of America 112:2314.

Lenhardt, M.L. 1994. Auditory behavior of the loggerhead sea turtle (*Caretta caretta*). Page 89. In: K.A. Bjorndahl, A.B. Bolten, D.A. Johnson, and P.J. Eliazar (compilers), Proceedings of the 14th Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFC-351.

Lettevall, E., C. Richter, N. Jaquet, E. Slooten, S. Dawson, H. Whitehead, J. Christal, and P. M. Howard. 2002. Social structure and residency in aggregations of male sperm whales. The Canadian Journal of Zoology 80:1189-1196.

Lobel, P. 2002. Diversity of fish courtship and spawning sounds and application for monitoring reproduction. The Journal of the Acoustical Society of America 112:1.

Lord, A., J. R. Waas, J. Innes, and M. J. Whittingham. 2001. Effects of human approaches to nests of northern New Zealand dotterels. Biological Conservation 98:233-240.

Lu, Z., A. N. Popper, and R. R. Fay. 1996. Behavioral detection of acoustic particle motion by a teleost fish (*Astronotus ocellatus*): sensitivity and directionality. Journal of Comparative Physiology A 179:227-233.

Luksenburg J.A. and E.C.M. Parsons. 2009. The effects of aircraft on cetaceans: implications for aerial whalewatching. SC/61/WW2. Available at: http://www.iwcoffice.org/_documents/sci_com/SC61docs/SC-61-WW2.pdf

Lynch, J. F. 2001. Marine mammals and low-frequency sound: progress since 1994. The Journal of the Acoustical Society of America 110:649.

Lynn, S. K. and D. L. Reiss. 1992. Pulse sequence and whistle production by two captive beaked whales, *Mesoplodon* species. Marine Mammal Science 8:299-305.

Lyrholm, T., O. Leimar, B. Johanneson, and U. Gyllensten. 1999. Sex-biased dispersal in sperm whales: contrasting mitochondrial and nuclear genetic structure of global populations. Proceedings of the Royal Society of London, Series B Biological Sciences 266:347-354.

Macleod, C. D. 2000. Review of the distribution of *Mesoplodon* species (order Cetacea, family Ziphiidae) in the North Atlantic. Mammal Review 30:1-8.

Madsen, P. T., D. A. Carder, K. Bedholm, and S. H. Ridgway. 2005. Porpoise clicks from a sperm whale nose - convergent evolution of 130 kHz pulses in toothed whale sonars? Bioacoustics 15:12.

Madsen, P. T., M. Johnson, N. A. de Soto, W. M. X. Zimmer, and P. Tyack. 2005. Biosonar performance of foraging beaked whales (*Mesoplodon densirostris*). The journal of Experimental Biology 208:181-194.

Madsen, P. T., M. Johnson, P. L. Tyack, N. Aguilar de Soto, and W. M. X. Zimmer. 2004. Biosonar performance of foraging Blainvilles beaked whales (Mesoplodon densirostris). The Journal of the Acoustical Society of America 115:2406.

Madsen, P. T., I. Kerr, and R. Payne. 2004. Echolocation clicks of two free-ranging, oceanic delphinids with different food preferences: false killer whales Pseudorca crassidens and Risso's dolphins Grampus griseus. The journal of Experimental Biology 207:1811-1823.

Madsen, P. T. and B. Mohl. 2000. Sperm whales (*Physeter catodon* L 1758) do not react to sounds from detonators. The Journal of the Acoustical Society of America 107:668-671.

Madsen, P. T., R. Payne, N. U. Kristiansen, M. Wahlberg, I. Kerr, and B. Mohl. 2002. Sperm whale sound production studied with ultrasound time/depth-recording tags. The journal of Experimental Biology 205:1899-1906.

Madsen, P. T., M. Wahlberg, and B. Mohl. 2002. Male sperm whale (*Physeter macrocephalus*) acoustics in a high-latitude habitat: Implications for echolocation and communication. Behavioral Ecology and Sociobiology 53:31-41.

Malakoff, D. 2001. A roaring debate over ocean noise. Science 291:576-578.

Malakoff, D. 2002. Suit ties whale deaths to research cruise. Pages 722-723 Science (Washington D C).

Malik, S., M. W. Brown, S. D. Kraus, and B. N. White. 2000. Analysis of mitochondrial DNA diversity within and between North and South Atlantic right whales. Marine Mammal Science 16:14.

Mann, D. A., Z. Lu, M. C. Hastings, and A. N. Popper. 1998. Detection of ultrasonic tones and simulated dolphin echolocation clicks by a teleost fish, the American shad (*Alosa sapidissima*). The Journal of the Acoustical Society of America 104:7.

Mann, J. 2000. Unraveling the dynamics of social life: long-term studies and observational methods. Pages 45-64 *in* J. Mann, R. C. Connor, P. L. Tyack, and H. Whitehead, editors. Cetacean societies. Field studies of dolphins and whales. University of Chicago Press, Chicago, Illinois.

Mann, J., R. C. Connor, P. L. Tyack, and H. Whitehead, editors. 2000. Ceracean societies. Field studies of dolphins and whales. University of Chicago Press, Chicago, Illinois.

Mansour, A. A. H., D. W. McKay, J. Lien, J. C. Orr, J. H. Banoub, N. Oien, and G. Stenson. 2002. Determination of pregnancy status from blubber samples in minke whales (Balaenoptera acutorostrata). Marine Mammal Science 18:9.

Marini, L., C. Consiglio, B. Catalano, T. Valentini, and G. Villetti. 1996. Aerial behavior in fin whales (*Balaenoptera physalus*) in the Mediterranean Sea. Marine Mammal Science 12:7.

Martensson, P. E., A. R. Lager Gotaas, E. S. Nordoy, and A. S. Blix. 1996. Seasonal changes in energy density of prey of Northeast Atlantic seals and whales. Marine Mammal Science 12:6.

Martin, A. R. and T. G. Smith. 1999. Strategy and capability of wild belugas, *Delphinapterus leucas*, during deep, benthic diving. The Canadian Journal of Zoology 77:1783-1793.

Martin, B., J. MacDonnell, N. E. Chorney, and D. Zeddies. 2012. Sound source verification of Fugro Geotechnical Sources: Final report: Boomer, sub-bottom profiler, multibeam sonar, and the R/V Taku. JASCO.

Marvit, P. and J. D. Crawford. 2000. Auditory discrimination in a sound-producing electric fish (Pollimyrus): Tone frequency and click-rate difference detection. The Journal of the Acoustical Society of America 108:7.

Marvit, P. and J. D. Crawford. 2000. Auditory thresholds in a sound-producing electric fish (Pollimyrus): Behavioral measurements of sensitivity to tones and click trains. The Journal of the Acoustical Society of America 107:6.

Mast, T. D., L. M. Hinkelman, M. J. Orr, and R. C. Waag. 1998. The effect of abdominal wall morphology on ultrasonic pulse distortion. Part II. Simulations. The Journal of the Acoustical Society of America 104:3651.

Mate, B. R., B. A. Lagerquist, and J. Calambokidis. 1999. Movements of North Pacific blue whales during the feeding season off southern California and their southern fall migration. Marine Mammal Science 15:12.

Matsuoka, K., P. Ensor, T. Hakamada, H. Shimada, S. Nishiwaki, F. Kasamatsu, and H. Kato. 2003. Overview of minke whale sightings surveys conducted on IWC/IDCR and SOWER Antarctic cruises from 1978/79 to 2000/01. Journal of Cetacean Research and Management 5: 2, 173-201. .

Matthews, J. N., S. Brown, D. Gillespie, M. Johnson, R. McLanaghan, A. Moscrop, D. Nowacek, R. Leaper, T. Lewis, and P. Tyack. 2001. Vocalisation rates of the North Atlantic right whale (*Eubalaena glacialis*). Journal of Cetacean Research and Management 3.

Matthews, J. N., L. Steiner, and J. Gordon. 2001. Mark-recapture analysis of sperm whale (*Physeter macrocephalus*) photo-id data from the Azores (1987-1995). Journal of Cetacean Research and Management 3.

Maybaum, H. L. 1990. Effects of a 3.3 kHz sonar system on humpback whales, *Megaptera novaeangliae*, in Hawaiian waters. EOS 71:92.

Maybaum, H. L. 1993. Responses of humpback whales to sonar sounds. The Journal of the Acoustical Society of America 94:1848-1849.

McCarthy, S. 2002. Do we kill whales with sound? Discover 23:60-65.

McCauley, R. D. and D. H. Cato. 2001. The underwater noise of vessels in the Hervey Bay (Queensland) whale watch fleet and its impact on humpback whales. The Journal of the Acoustical Society of America 109:2455.

McCauley, R. D., J. Fewtrell, A. J. Duncan, C. Jenner, M.-N. Jenner, J. D. Penrose, R. I. T. Prince, A. Adhitya, J. Murdoch and K. McCabe 2000b. Marine seismic surveys: analysis and propagation of air-gun signals; and effects of air-gun exposure on humpback whales, sea turtles, fishes and squid. Prepared for the Australian Petroleum Production Exploration Association by the Centre for Marine Science and Technology, Project CMST 163, Report R99-15. 203p.

McCauley, R. D., M.-N. Jenner, C. Jenner, K. A. McCabe and J. Murdoch. 1998. The response of humpback whales (*Megaptera novaeangliae*) to offshore seismic survey noise: preliminary results of observations about a working seismic vessel and experimental exposures. APPEA Journal 38:692-707.

McCauley, R. D., J. Fewtrell, and A. N. Popper. 2003. High intensity anthropogenic sound damages fish ears. The Journal of the Acoustical Society of America 113:638-642.

McCauley, R. D., C. Jenner, J. L. Bannister, and D. H. Cato. 2001. Blue whale calling off the Western Australian coast. The Journal of the Acoustical Society of America 109:2391.

McClatchie, S., G. Macaulay, R. F. Coombs, P. Grimes, and A. Hart. 1999. Target strength of an oily deep-water fish, orange roughy (*Hoplostethus atlanticus*). I. Experiments. The Journal of the Acoustical Society of America 106:12.

McClatchie, S. and Z. Ye. 2000. Target strength of an oily deep-water fish, orange roughy (*Hoplostethus atlanticus*). II. Modeling. The Journal of the Acoustical Society of America 107:6.

McDonald, M. A. and C. G. Fox. 1999. Passive acoustic methods applied to fin whale population density estimation. The Journal of the Acoustical Society of America 105:2643.

McDonald, M. A., J. A. Hildebrand, and S. C. Webb. 1995. Blue and fin whales observed on a seafloor array in the northeast Pacific. The Journal of the Acoustical Society of America 98:712-721.

McDonald, M. A. and S. E. Moore. 2002. Calls recorded from North Pacific right whales (*Eubalaena japonica*). Journal of Cetacean Research and Management 4.

McKibben, J. R. and A. H. Bass. 1998. Behavioral assessment of acoustic parameters relevant to signal recognition and preference in a vocal fish. The Journal of the Acoustical Society of America 104:14.

McMahon, C. R., H. R. Burton, and M. N. Bester. 2003. A demographic comparison of two southern elephant seal populations. The Journal of Animal Ecology 72:61-74.

Melcón, M. L., A. J. Cummins, S. M. Kerosky, L. K. Roche, S. M. Wiggins, and J. A. Hildebrand. 2012. Blue whales respond to anthropogenic noise. PLoS ONE 7:e32681.

Mellinger, D. K. and C. W. Clark. 2000. Recognizing transient low-frequency whale sounds by spectrogram correlation. The Journal of the Acoustical Society of America 107:3518.

Mellinger, D. K., K. Stafford, C. G. Fox, and S. E. Moore. 2002. Automatic detection of sperm whale sounds from the Gulf of Alaska. The Journal of the Acoustical Society of America 112:2399.

Mellinger, D. K., K. M. Stafford, and C. G. Fox. 2004. Seasonal occurrence of sperm whale (*Physeter macrocephalus*) sounds in the Gulf of Alaska, 1999-2001. Marine Mammal Science 20:15.

Mercado III, E. and L. N. Frazer. 1999. Environmental constraints on sound transmission by humpback whales. The Journal of the Acoustical Society of America 106:3004.

Miller, C. A., D. Reeb, P. B. Best, A. R. Knowlton, M. W. Brown and M. J. Moore. 2011. Blubber thickness in right whales *Eubalaena glacialis* and *Eubalaena australis* related with reproduction, life history status and prey abundance. Marine Ecology Progress Series 438: 267-283.

Miller, P. J. O. 2002. Mixed-directionality of killer whale stereotyped calls: A direction of movement cue? Behavioral Ecology and Sociobiology 52:262-270.

Miller, P. J. O. and D. E. Bain. 2000. Within-pod variation in the sound production of a pod of killer whales, *Orcinus orca*. Animal Behaviour 60:617-628.

Miller, P. J. O., N. Biassoni, A. Samuels, and P. L. Tyack. 2000. Whale songs lengthen in response to sonar. Page 903 Nature (London).

Mitchell, E. 1974. Present status of northwest Atlantic fin and other whale stocks. Pages 108-169 The Whale Problem: A Status Report. Harvard University Press, Cambridge, Massachusetts.

Mitchell, E. and V. M. Kozicki. 1984. Reproductive condition of male sperm whales, *Physeter macrocephalus*, taken off Nova Scotia. Rep. int Whal. Commn. (Special Issue 6):243-252.

Mitson, R. B. 1995. Underwater noise of research vessels. Review and recommendations. Cooperative Research Report No. 209, International Council for the Exploration of the Sea, Copenhagen, Denmark.

Miyakita, T., P. A. Hellstrom, E. Frimanson, and A. Axelsson. 1992. Effect of low level acoustic stimulation on temporary threshold shift in young humans. Hearing research 60:149-155.

Moberg, G. P. 2000. Biological response to stress: implications for animal welfare. Pages 1 - 21 *in* G. P. Moberg and J. A. Mench, editors. The biology of animal stress. Basic principles and implications for animal welfare. Oxford University Press, Oxford, United Kingdom.

Mobley, J. R., Jr., L. M. Herman, and A. S. Frankel. 1988. Responses of wintering humpback whales *Megaptera noveaeangliae* to playback of recordings of winter and summer vocalizations and of synthetic sound. Behavioral Ecology and Sociobiology 23:211-224.

Mohl, B. 2001. Sound transmission in the nose of the sperm whale *Physeter catodon*. A post mortem study. Journal of Comparative Physiology A Sensory Neural and Behavioral Physiology 187:335-340.

Mohl, B., M. Wahlberg, P. T. Madsen, A. Heerfordt, and A. Lund. 2003. The monopulsed nature of sperm whale clicks. The Journal of the Acoustical Society of America 114:12.

Montie, E.W., R.J. Letcher, C.M. Reddy, M.J. Moore, B. Rubinstein and M.E. Hahn. 2010. Brominated flame retardants and organochlorine contaminants in winter flounder, harp and hooded seals, and North Atlantic right whales from the Northwest Atlantic Ocean. Marine Pollution Bulletin 60(8):1160-9.

Moore, K. E., W. A. Watkins, and P. L. Tyack. 1993. Pattern similarity in shared codas from sperm whales (*Physeter catodon*). Marine Mammal Science 9:1-9.

Moore, M. J., S. D. Berrow, B. A. Jensen, P. Carr, R. Sears, V. J. Rowntree, R. Payne, and P. K. Hamilton. 1999. Relative abundance of large whales around South Georgia (1979-1998). Marine Mammal Science 15:16.

Moore, M. J., C. A. Miller, M. S. Morss, R. Arthur, W. A. Lange, K. G. Prada, M. K. Marx, and E. A. Frey. 2001. Ultrasonic measurement of blubber thickness in right whales. Journal of Cetacean Research and Management Supplement 2:301 - 309.

Mossbridge, J. A. and J. A. Thomas. 2001. An "acoustic niche" for Antarctic killer whale and leopard seal sounds. The Journal of the Acoustical Society of America 109:2390.

Muchnik, C., E. Sahartov, E. Peleg, and M. Hildesheimer. 1992. Temporary threshold shift due to noise exposure in guinea pigs under emotional stress. Hearing research 58:101-106.

Mullin, K. D., T. A. Jefferson, L. J. Hansen, and W. Hoggard. 1994. First sightings of melon-headed whales (*Peponocephala electra*) in the Gulf of Mexico. Marine Mammal Science 10:342.

Murphy, W. J. and L. E. van Campen. 2001. Temporary threshold shift in ABRs and DPOAEs following noise exposure in Long-Evans rats. The Journal of the Acoustical Society of America 109:2373.

Murray, S. O., E. Mercado, and H. L. Roitblat. 1998. Characterizing the graded structure of false killer whale (*Pseudorca crassidens*) vocalizations. The Journal of the Acoustical Society of America 104:1679.

Murray, S. O., E. Mercado, and H. L. Roitblat. 1998. The neural network classification of false killer whale (*Pseudorca crassidens*) vocalizations. The Journal of the Acoustical Society of America 104:3626.

Mutlu, E. 2000. Detection of harbor porpoises and white whales (Beluga) sound using the high frequency bioacoustics and their echo structures. The Journal of the Acoustical Society of America 108:2584.

Nachtigall, P. E., J. L. Pawloski, and W. W. L. Au. 2003. Temporary threshold shifts and recovery following noise exposure in the Atlantic bottlenosed dolphin (*Tursiops truncatus*). The Journal of the Acoustical Society of America 113:5.

Nachtigall, P. E., A. Y. Supin, J. Pawloski, and W. W. L. Au. 2004. Temporary threshold shifts after noise exposure in the bottlenose dolphin (*Tursiops truncatus*) measured using evoked auditory potentials. Marine Mammal Science 20:15.

Nachtigall, P. E., M. M. L. Yuen, T. A. Mooney, and K. A. Taylor. 2005. Hearing measurements from a stranded infant Risso's dolphin, *Grampus griseus*. The journal of Experimental Biology 208:4181.

National Marine Fisheries Service (NMFS). 1998. Recovery Plan for the Shortnose Sturgeon (*Acipenser brevirostrum*). Prepared by the Shortnose Sturgeon Recovery Team for the National Marine Fisheries Service, Silver Spring, Maryland. 104p.

NMFS. 2000. Status Review of Smalltooth Sawfish (*Pristis pectinata*). December 2000.

NMFS 2006. Biological Opinion on the Funding and Permitting of Seismic Surveys by the National Science Foundation and the National Marine Fisheries Service in the Eastern Tropical Pacific Ocean from March to April 2006. National Marine Fisheries Service, Silver Spring, Maryland. 76p.NMFS. 2010. Recovery plan for the fin whale (Balaenoptera physalus). National Marine Fisheries Service, Silver Spring, MD. 121 pp.

NMFS. 2012. Reinitiation of Endangered Species Act (ESA) Section 7 Consultation on the Continued Implementation of the Sea Turtle Conservation Regulations, as Proposed to Be Amended, and the Continued Authorization of the Southeast U.S. Shrimp Fisheries in Federal Waters under the Magnuson-Stevens Act. NMFS SER. May 8, 2012.

National Ocean Service. 2003. Oil and sea turtles, biology, planning and response. S. Shigenake (ed.) National Oceanic and Atmospheric Administration, National Ocean Service, Office of Response and Restoration. 111 pp.

National Research Council. 1990. Decline of the sea turtles: causes and prevention. National Academy Press, Washington, D.C.

National Research Council. 1995. Science and the Endangered Species Act. National Academy Press, Washington, D.C.

National Research Council. 2000. Marine mammals and low-frequency sound: progress since 1994. National Academy Press, Washington, D.C.

National Research Council. 2003. Ocean noise and marine mammals. National Academies Press, Washington, D.C.

National Research Council. 2005. Marine mammal populations and ocean noise : determining when noise causes biologically significant effects. National Academies Press, Washington, D.C.

National Science Foundation and U.S. Dept. of the Interior, U.S. Geological Survey. 2011. Final Programmatic Environmental Impact Statement/Overseas Environmental Impact Statement for Marine Seismic Research Funded by the National Science Foundation or conducted by the U.S.

Geological Survey. June 2011. 514 pp. Available at: http://www.nsf.gov/geo/oce/envcomp/usgs-nsf-marine-seismic-research/nsf-usgs-final-eis-oeis_3june2011.pdf.

Nawojchik, R., D. J. S. Aubin, and A. Johnson. 2003. Movements and dive behavior of two stranded, rehabilitated long-finned pilot whales (*Globicephala melas*) in the Northwest Atlantic. Marine Mammal Science 19:8.

Neff, J.M., S. McKelvie and R.C. Ayers, Jr. 2000. Environmental impacts of synthetic based drilling fluids. Report prepared for MMS by Robert Ayers & Associates, Inc. August 2000. U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, LA. OCS Study MMS 2000-064. 118 pp.

Nelson M, Garron M, Merrick RL *et al.* 2007. Mortality and Serious Injury Determinations for Baleen Whale Stocks along the United States Eastern Seaboard and Adjacent Canadian Maritimes, 2001-2005. In: *Northeast Fisheries Science Center Reference Document 07- 05*. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Science Center, Woods Hole, MA.

Nero, R. W., C. H. Thompson, and R. H. Love. 2002. Fish schools as potential clutter and false targets: Observations on the New Jersey shelf. The Journal of the Acoustical Society of America 112:1.

Newcomb, J., R. Fisher, R. Field, G. Rayborn, S. Kuczaj, G. Ioup, J. Ioup, and A. Turgut. 2002. Measurements of ambient noise and sperm whale vocalizations in the northern Gulf of Mexico using near bottom hydrophones. Ocean 3:1365-1371.

Newcomb, J. J., A. J. Wright, S. Kuczaj, R. Thames, W. R. Hillstrom, and R. Goodman. 2004. Underwater ambient noise and sperm whale click detection during extreme wind speed conditions. AIP Conference Proceedings 728:296-306.

Niklitschek E. S. and D. H. Secor. 2009. Dissolved oxygen temperature and salinity effects on the ecophysiology and survival of juvenile Atlantic sturgeon in estuarine waters. J. Exp. Mar. Biol. Ecol. 381: Suppl. 1: 150-172.

Niklitschek, E. J. and D.H. Secor. 2005. Modeling spatial and temporal variation of suitable nursery habitats for Atlantic sturgeon in the Chesapeake Bay. Estuarine, Coastal and Shelf Science 64 135-148.

Niklitschek, E.S. and D.H. Secor. 2010. Experimental and field evidence of behavioral habitat selection by juvenile Atlantic (*Acipenser oxyrinchus*) and shortnose (*Acipenser brevirostrum*) sturgeons. J. Fish Biol. 77: 1293-1308.

Nolan, C. P., G. M. Liddle, and J. Elliot. 2000. Interactions between killer whales (*Orcinus orca*) and sperm whales (*Physeter macrocephalus*) with a longline fishing vessel. Marine Mammal Science 16:6.

Normandeau Associates, Inc. 2012. Effects of noise on fish, fisheries, and invertebrates in the U.S. Atlantic and Arctic from energy industry sound-generating activities. A literature synthesis for the U.S. Department of Interior, Bureau of Ocean Energy Management. Contract # M11PC00031. 153 pp.

Normile, D. and J. t. Mervis. 2000. Japan's whaling program carries heavy baggage. Pages 2264-2265 Science (Washington D C).

Norris, S. 2002. Creatures of culture? Making the case for cultural systems in whales and dolphins. Bioscience 52:9-14.

Norris, T. F. 1994. Effects of boat noise on the acoustic behavior of humpback whales. The Journal of the Acoustical Society of America 96:3251.

Norris, T. F., M. M. Donald, and J. Barlow. 1999. Acoustic detections of singing humpback whales (*Megaptera novaeangliae*) in the eastern North Pacific during their northbound migration. The Journal of the Acoustical Society of America 106:9.

Norton, S. B., S. B. Cormier, G. W. Suter II, B. Subramanian, E. Lin, D. Altfater, and B. Counts. 2002. Determining probable causes of ecological impairment in the Little Scioto River, Ohio, USA: Part 1. listing candidate causes and analyzing evidence. Environmental Toxicology and Chemistry 21:1112-1124.

Notarbartolo-di-Sciara, G., M. Zanardelli, M. Jahoda, S. Panigada, and S. Airoldi. 2003. The fin whale *Balaenoptera physalus* (L. 1758) in the Mediterranean Sea. Mammal Review 33:105-150.

Nottestad, L. and B. E. Axelsen. 1999. Herring schooling manoeuvres in response to killer whale attacks. The Canadian Journal of Zoology 77:1540-1546.

Nowacek, D. P., M. P. Johnson, and P. L. Tyack. 2004. North Atlantic right whales (*Eubalaena glacialis*) ignore ships but respond to alerting stimuli. Proceedings of the Royal Society Biological Sciences Series B 271:227-231.

Nowacek, D.P, L.H. Thorne, D.W. Johnston and P.L. Tyack. 2007. Reponses of cetaeans to anthropogenic noise. Mammalian Review, 37(2), 81-115.

O'Brien, W. D., Jr, L. A. Frizzell, R. M. Weigel, and J. F. Zachary. 2000. Ultrasound-induced lung hemorrhage is not caused by inertial cavitation. The Journal of the Acoustical Society of America 108:1290.

Ohsumi, S., and S. Wada. 1974. Status of whale stocks in the North Pacific, 1972. Reports of the International Whaling Commission 24:114-126.

Oleson, E. M., J. Barlow, J. Gordon, S. Rankin, and J. A. Hildebrand. 2003. Low frequency calls of Bryde's whales. Marine Mammal Science 19:406.

Olson, T. L., B. K. Gilbert, and R. C. Squibb. 1997. The effects of increasing human activity on brown bear use of an Alaskan River. Biological Conservation 82:95-99.

Ona, E. and I. Svellingen. 2001. Target strength analysis in high fish densities using a probing split beam transducer. The Journal of the Acoustical Society of America 109:2305.

Orians, G., F. Briand, J. Diamond, T. Colborn, E. Gomez, R. Guillemin, A. Klug, M. Konishi, J. Lubchenco, L. Mee, E. Norse, G. N. Di Sciara, R. Payne, C. Safina, D. Suzuki, E. O. Wilson, and G. t. Woodwell. 2003. "Scientists versus Whaling": Whose errors of judgment? Bioscience 53:200; 202-203.

Palacios, D. M. and B. R. Mate. 1996. Attack by false killer whales (*Pseudorca crassidens*) on sperm whales (*Physeter macrocephalus*) in the Galapagos Islands. Marine Mammal Science 12:6.

Palka, D.L. 2006. Summer abundance estimates of cetaceans in US North Atlantic Navy Operating Areas. Northeast Fish. Sci. Cent. Ref. Doc. 06-03; 41 p.

Parks, S. E. 2003. Response of North Atlantic right whales (*Eubalaena glacialis*) to playback of calls recorded from surface active groups in both the North and South Atlantic. Marine Mammal Science 19:18.

Parks, S. E., P. K. Hamilton, S. D. Kraus, and P. L. Tyack. 2005. The gunshot sound produced by male North Atlantic right whales (*Eubalaena glacialis*) and its potential function in reproductive advertisement. Marine Mammal Science 21:18.

Patenaude, N. J., C. S. Baker, and N. J. Gales. 1998. Observations of southern right whales on New Zealand's subantarctic wintering grounds. Marine Mammal Science 14:5.

Patenaude, N. J., W. J. Richardson, M. A. Smultea, W. R. Koski, G. W. Miller, B. Wursig, and C. R. Greene Jr. 2002. Aircraft sound and disturbance to bowhead and beluga whales during spring migration in the Alaskan beaufort Sea. Marine Mammal Science 18:309-335.

Patro, R., L. Zedel, and C. Spanu-Tollefsen. 2000. Monitoring fish movement using an ADCP. The Journal of the Acoustical Society of America 108:2489.

Pavan, G., T. J. Hayward, J. F. Borsani, M. Priano, M. Manghi, C. Fossati, and J. Gordon. 2000. Time patterns of sperm whale codas recorded in the Mediterranean Sea 1985-1996. The Journal of the Acoustical Society of America 107:3487.

Payne, R. 1986. Long term behavioral studies of the southern right whale (Eubalaena australis). Report of the International Whaling Commission Special Issue 10:161-167.-Right Whales Past and Present status. Proceedings of the Workshop on the status of Right Whales. Robert L. Brownell, Peter B. Best, John H. Prescott-Eds.).

Payne, P. M., and coauthors. 1990. Recent fluctuations in the abundance of baleen whales in the southern Gulf of Maine in relation to changes in prey abundance. Fishery Bulletin 88(4):687-696.

Pearson, W. H., J. R. Skalski and C. I. Malme 1992. Effects of sounds from a geophysical survey device on behavior of captive rockfish (*Sebastes* spp.). Canadian Journal of Fisheries and Aquatic Sciences 49:1343-1356.

Pedersen, B. and M. V. Trevorrow. 1999. Continuous monitoring of fish in a shallow channel using a fixed horizontal sonar. The Journal of the Acoustical Society of America 105:10.

Perry, S. L., D. P. DeMaster, and G. K. Silber. 1999a. The Great Whales: History and Status of Six Species Listed as Endangered Under the U.S. Endangered Species Act of 1973. Marine Fisheries Review 61(1):1-74.

Pershing, A. J., E. H. J. Head, C. H. Greene, and J. W. Jossi. 2010. Pattern and scale of variability among Northwest Atlantic Shelf plankton communities. Journal of Plankton Research 32(12):1661-1674.

Peterson, D. L., M. B. Bain and N. Haley. 2000. Evidence of declining recruitment of Atlantic sturgeon in the Hudson River. North American Journal of Fisheries Management. 20:231-238.

Peterson, D.L., P. Schueller, R. DeVries, J. Fleming, C. Grunwald, I. Wirgin. 2008. Annual run size and genetic characteristics of Atlantic sturgeon in the Altamaha River. Transactions of the American Fisheries Society 137:393-401.

Piantadosi, C. A. and E. D. Thalmann. 2004. Pathology: Whales, sonar and decompression sickness. Nature 428:n.

Pietraszek, J. and S. Atkinson. 1994. Concentrations of estrone sulfate and progesterone in plasma and saliva, vaginal cytology, and bioelectric impedance during the estrous cycle of the Hawaiian monk seal (*Monachus schauinslandi*). Marine Mammal Science 10:430.

Pike, D. and J.C. Stiner. 2007. Sea turtle species vary in their susceptibility to tropical cyclones. Oecologia, 153:471–478.

Pirotta, E., R. Milor, N. Quick, D. Moretti, N. Di Marzio, P. Tyack, I. Boyd, and G. Hastie. 2012. Vessel noise affects beaked whale behavior: results of a dedicated acoustic response study. PLoS ONE 7:e42535.

Pitman, R. L., L. T. Ballance, S. L. Mesnick, and S. J. Chivers. 2001. Killer whale predation on sperm whales: observations and implications. Marine Mammal Science 17:494.

Polyakov, I.V., C.A. Alexeev, U.S. Bhatt, E.I. Polyakova and X. Zhang. 2010. North Atlantic warming: patterns of long-term trend and multidecadal variability. Climate Dynamics, 34:439-457.

Ponganis, P. J., G. L. Kooyman, and M. A. Castellini. 1995. Multiple sightings of Arnoux's beaked whales along the Victoria Land coast. Marine Mammal Science 11:4.

Popper, A.N. and M. C. Hastings. 2009. The effects of human-generated sounds on fish. *Integrative Zoology* 2009; 4: 43-52.

Popper, A. N., R. D. McCauley, and J. Fewtrell. 2002. Impact of anthropogenic sounds on fishes. The Journal of the Acoustical Society of America 112:2431.

Popper, A. N., M. E. Smith, P. A. Cott, B. W. Hanna, A. O. Macgillivray, M. E. Austin, and D. A. Mann. 2005. Effects of exposure to seismic airgun use on hearing of three fish species. The Journal of the Acoustical Society of America 117:3958.

Postuma, F. A. and Gasalla, M. A. 2010. On the relationship between squid and the environment: artisanal jigging for ***Loligo plei*** at São Sebastião Island (24°S), southeastern Brazil. – ICES Journal of Marine Science, 67: 1353–1362.

Pyzik, L., J. Caddick, and P. Marx. 2004. Chesapeake Bay: introduction to an ecosystem. Chesapeake Bay Program, EPA Publication 903-R-04-003. Annapolis, Maryland

Rankin, S., D. Ljungblad, C. Clark, and K. Hiderhiro. 2005. Vocalisations of Antarctic blue whales, *Balaenoptera musculus intermedia*, recorded during the 2001/2002 and 2002/2003 IWC/SOWER circumpolar cruises, Area V, Antarctica. Journal of Cetacean Research and Management 7:13-20.

Ratilal, P. and N. C. Makris. 2002. Coherent versus diffuse surface and volume reverberation in an ocean wave guide: Reverberation rings, modal decoupling, and possible fish scattering in Geoclutter 2001. The Journal of the Acoustical Society of America 112:2280.

Read, A. J. and P. R. Wade. 2000. Status of marine mammals in the United States. Conservation Biology 14:929-940.

Reeder, D. B., T. K. Stanton, D. Chu, and J. M. Jech. 2000. Broadband acoustic backscattering from alewife fish: Experiment and analysis. The Journal of the Acoustical Society of America 108:2457.

Reeves, R. R. 1992. Whale responses to anthropogenic sounds: a literature review. Science and Research Series No. 47, New Zealand Department of Conservation, Wellington, New Zealand.

Reeves, R. R., B. D. Smith, E. Crespo, and G. Notobartolo di Sciara. 2003. Dolphins, whales and porpoises: 2002-2010 Conservation action plan for the world's cetaceans. IUCN - The World Conservation Union, Gland, Switzerland.

Reeves, R. R., T. D. Smith, E. A. Josephson, P. J. Clapham, and G. Woolmer. 2004. Historical observations of humpback and blue whales in the North Atlantic Ocean: Cities to migratory routes and possibly additional feeding grounds. Marine Mammal Science 20:13.

Reid, K., A. S. Brierley, and G. A. Nevitt. 2000. An initial examination of relationships between the distribution of whales and Antarctic krill Euphausia superba at South Georgia. Journal of Cetacean Research and Management 2.

Reif, J. S., A. Bachand, A. A. Aguirre, D. L. Borjesson, L. Kashinsky, R. Braun, and G. Antonelis. 2004. Morphometry, hematology, and serum chemistry in the Hawaiian monk seal (*Monachus schauinslandi*). Marine Mammal Science 20:10.

Rendell, L. and H. Whitehead. 2004. Do sperm whales share coda vocalizations? - Insights into coda usage from acoustic size measurement. Animal Behaviour 67:865-874.

Rendell, L. and H. Whitehead. 2005. Coda playbacks to sperm whales in Chilean waters. Marine Mammal Science 21:10.

Rendell, L., H. Whitehead, and A. Coakes. 2005. Do breeding male sperm whales show preferences among vocal clans of females? Marine Mammal Science 21:6.

Rendell, L. E. and J. C. D. Gordon. 1999. Vocal response of long-finned pilot whales (*Globicephala melas*) to military sonar in the Ligurian Sea. Marine Mammal Science 15:198-204.

Rendell, L. E., J. N. Matthews, A. Gill, J. C. D. Gordon, and D. W. Macdonald. 1999. Quantitative analysis of tonal calls from five odontocete species, examining interspecific and intraspecific variation. Journal of Zoology 249:403-410.

Rendell, L. E. and H. Whitehead. 2003. Comparing repertoires of sperm whale codas: a multiple methods approach. Bioacoustics 14:61 - 81.

Rendell, L. E. and H. Whitehead. 2003. Vocal clans in sperm whales (*Physeter macrocephalus*). Proceedings of the Royal Society Biological Sciences Series B 270:225-231.

Richardson, W. J., R. A. Davids, C. R. Evans, and P. Norton. 1985. Distribution of bowheads and industrial activity. Report prepared by LGL Ecological Research Associates, Inc. for U.S. Department of the Interior, Minerals Management Service, Bryan, Texas, and Reston, Virginia.

Richardson, W. J., K. J. Finley, G. W. Miller, and R. A. Davis. 1995. Feeding, social and migration behavior of bowhead whales, *Balaena mysticetus*, in Baffin Bay vs the Beaufort Sea - regions with different amounts of human activity. Marine Mammal Science 11:1.

Richardson, W. J., C. R. Greene Jr, C. I. Malme, and D. H. Thomson. 1995b. Marine mammals and noise. Academic Press, Inc., San Diego, California.

Richardson, W. J., R. S. Wells, and B. Würsig. 1985. Disturbance responses of bowheads, 1980-84. Pages 89-196 *in* W. J. Richardson, editor. Behavior, disturbance and distribution of bowhead whales *Balaena mysticetus* in the eastern Beaufort Sea, 1980-84. Report from LGL Ecological Research Associates, Inc. for U.S. Minerals Management Service, Bryan, Texas, and Reston, Virginia.

Richardson, W. J., B. Würsig, and C. R. Greene Jr. 1990. Reactions of bowhead whales, *Balaena mysticetus*, to drilling and dredging noise in the Canadian Beaufort Sea. Marine Environmental Research 29:135-160.

Richardson, W. J., B. Würsig, and C. R. Greene Jr. 1986. Reactions of bowhead whales, *Balaena mysticetus*, to seismic exploration in the Canadian Beaufort Sea. The Journal of the Acoustical Society of America 79:1117-1128.

Richter, C., S.M. Dawson and E. Slooten. 2003. Sperm whale watching off Kaikoura, New Zealand: effects of current activities on surfacing and vocalization patterns. Science for Conservation 219. New Zealand Department of Conservation; Wellington, New Zealand.

Richter, C., S. Dawson, and E. Slooten. 2006. Impacts of commercial whale watching on male sperm whales at Kaikoura, New Zealand. Marine Mammal Science 22:46-63.

Ridgway, S. H., D. A. Carder, T. Kamolnick, R. R. Smith, C. E. Schlundt, and W. R. Elsberry. 2001. Hearing and whistling in the deep sea: depth influences whistle spectra but does not attenuate hearing by white whales (*Delphinapterus leucas*) (Odontoceti, Cetacea). The journal of Experimental Biology 204:3829-3841.

Ridgway, S. H., E. G. Wever, J. G. McCormick, J. Palin and J. H. Anderson 1969. Hearing in the giant sea turtle, *Chelonia mydas*. Proceedings of the National Academy of Sciences of the United States of America 64:884-890.

Ridoux, V., C. Guinet, C. Liret, P. Creton, R. Steenstrup, and G. Beauplet. 1997. A video sonar as a new tool to study marine mammals in the wild: Measurements of dolphin swimming speed. Marine Mammal Science 13:11.

Robbins, M. W. 2003. Will the Navy's new sonar harm whales? Page 75 Discover (New York).

Robineau, D. and M. Vely. 1993. Stranding of a specimen of Gervais' beaked whale (*Mesoplodon europaeus*) on the coast of West Africa (Mauritania). Marine Mammal Science 9:438-440.

Robinette, M. 2004. Effects of N-acetylcysteine on noise-induced temporary threshold shift and temporary emission shift. The Journal of the Acoustical Society of America 115:2388.

Robinson, C. 2002. Ten years of hydroacoustical observations of the behavior and distribution of small pelagic fish in the west coast of Baja California, Mexico. The Journal of the Acoustical Society of America 112:2207.

Romero, A., K. T. Hayford, and J. Romero. 2002. The marine mammals of Grenada, W.I., and their conservation status. Mammalia 66:479-494.

Rommel, S. A., A. M. Costidis, A. Fernandez, P. D. Jepson, D. A. Pabst, W. A. McLellan, D. S. Houser, T. W. Cranford, A. L. van Helden, D. M. Allen, and N. B. Barros. 2006. Elements of beaked whale anatomy and diving physiology, and some hypothetical causes of sonar-related stranding. Journal of Cetacean Research and Management 7:189 - 209.

Rooney, A. P., R. L. Honeycutt, S. K. Davis, and J. N. Derr. 1999. Evaluating a putative bottleneck in a population of bowhead whales from patterns of microsatellite diversity and genetic disequilibria. Journal of Molecular Evolution 49:682-690.

Rooney, A. P., R. L. Honeycutt, and J. N. Derr. 2001. Historical population size change of bowhead whales inferred from DNA sequence polymorphism data. Evolution 55:1678-1685.

Rossi, G., P. Solero, M. Rolando, and M. Olina. 1991. Recovery time of the temporary threshold shift for delayed evoked otoacoustic emissions and tone bursts. Journal for Oto-Rhino-Laryngology and its Borderlands 53:15-18.

Saegusa, A. t. 2000. Japan squares up to conservationists over grey whale's status. Pages 531-532 Nature (London).

Sakhalin Energy Investment Company Limited. 2002. Western gray whale protection plan: framework for monitoring and mitigation measures related to Sakhalin energy oil and gas operations on the northeast coast of Sakhalin Island, Russia. Unpublished report prepared by Sakhalin Energy Investment Company Limited, Yuzhno-Sakhalinsk, Russia.

Salden, D. R., L. M. Herman, M. Yamaguchi, and F. t. Sato. 1999. Multiple visits of individual humpback whales (*Megaptera novaeangliae*) between the Hawaiian and Japanese winter grounds. The Canadian Journal of Zoology 77:504-508.

Samuels, A. and P. L. Tyack. 2000. Flukeprints: a history of studying cetacean societies. Pages 9-44 *in* J. Mann, R. C. Connor, P. L. Tyack, and H. Whitehead, editors. Cetacean societies. Field studies of dolphins and whales. University of Chicago Press, Chicago, Illinois.

Schlundt, C. E., J. J. Finneran, D. A. Carder, and S. H. Ridgway. 2000. Temporary shift in masked hearing thresholds of bottlenose dolphins, *Tursiops truncatus*, and white whales, *Delphinapterus leucas*, after exposure to intense tones. The Journal of the Acoustical Society of America 107:3496.

Scholik, A. R. and H. Y. Yan. 2002. Effects of boat engine noise on the auditory sensitivity of the fathead minnow, *Pimephales promelas*. Environmental Biology of Fishes 63:7.

Scholz, N. L., N. K. Truelove, B. L. French, B. A. Berejikian, T. P. Quinn, E. Casillas, and T. K. Collier. 2000. Diazinon disrupts antipredator and homing behaviors in chinook salmon (*Oncorhynchus tshawytscha*). Canadian Journal of Fisheries and Aquatic Sciences 57:1911-1918.

Schrope, M. 2002. Whale deaths caused by US Navy's sonar. Page 106 Nature (London).

Schueller, P. and D. L. Peterson. 2010. Abundance and recruitment of juvenile Atlantic sturgeon in the Altamaha River, Georgia. Transactions of the American Fisheries Society 139:1526-1535.

Schusterman, R. J., D. Kastak, D. H. Levenson, C. J. Reichmuth, and B. L. Southall. 2000. Why pinnipeds don't echolocate. The Journal of the Acoustical Society of America 107:2256.

Scott, T. M. and S. Sadove. 1997. Sperm whale, *Physeter macrocephalus*, sightings in the shallow shelf waters off Long Island, New York. Marine Mammal Science 13:4.

Secor, D.H. 2002. Atlantic sturgeon fisheries and stock abundances during the late nineteenth century. Biology, Management, and Protection of North American Sturgeon. American Fisheries Society Symposium 28: 89-98.

Secor, D.H. and E.J. Niklitschek. 2002. Sensitivity of sturgeons to environmental hypoxia: A review of physiological and ecological evidence, p. 61-78 In: R.V. Thurston (Ed.) Fish Physiology, Toxicology, and Water Quality. Proceedings of the Sixth International Symposium, La Paz, MX, 22-26 Jan. 2001. U.S. Environmental Protection Agency Office of Research and Development, Ecosystems Research Division, Athens, GA. EPA/600/R-02/097. 372 pp.

Secor, D. H. and J. R. Waldman. 1999. Historical abundance of Delaware Bay Atlantic sturgeon and potential rate of recovery. American Fisheries Society Symposium 23:203-216.

Shelden, K. E. W., S. E. Moore, J. M. Waite, P. R. Wade, and D. J. Rugh. 2005. Historic and current habitat use by North Pacific right whales *Eubalaena japonica* in the Bering Sea and Gulf of Alaska. Mammal Review 35:129-155.

Sibly, R. M. and J. Hone. 2002. Population growth rate and its determinants: an overview. Philosophical Transactions of the Royal Society of London B Biological Sciences 357:1153-1170.

Sibly, R. M. and J. Hone. 2003. Population growth rate and its determinants: an overview. Pages 11 - 40 *in* R. M. Sibly, J. Hone, and T. H. Clutton-Brock, editors. Wildlife population growth rates. Cambridge University Press, Cambridge, United Kingdom.

Sibly, R. M., J. Hone, and T. H. Clutton-Brock, editors. 2003. Wildlife population growth rates. Cambridge University Press, Cambridge, United Kingdom.

Sidorovskaia, N. A., G. E. Ioup, J. W. Ioup, and J. W. Caruthers. 2004. Acoustic propagation studies for sperm whale phonation analysis during LADC experiments. AIP Conference Proceedings 728:288-295.

Sidorovskaia, N. A., I. A. Udovydchenkov, I. I. Rypina, G. E. Ioup, J. W. Ioup, J. W. Caruthers, J. Newcomb, and R. Fisher. 2004. Propagation modeling for sperm whale acoustic clicks in the northern Gulf of Mexico. The Journal of the Acoustical Society of America 115:2556.

Silber, G. K., J. Slutsky and S. Bettridge. 2010. Hydrodynamics of a ship/whale collision. Journal of Experimental Marine Biology and Ecology. 391(2010) 10-19.

Silvers, L. E., P. E. Rosel, and D. R. Salden. 2002. DNA sequence analysis of a North Pacific humpback whale (*Megaptera novaeangliae*) placenta. The Canadian Journal of Zoology 80:1141-1144.

Simard, Y., J. K. Horne, D. Lavoie, and I. McQuinn. 2000. Capelin TS: Effect of individual fish variability. The Journal of the Acoustical Society of America 108:2458.

Sims, D. W., M. J. Genner, A. J. Southward, and S. J. Hawkins. 2001. Timing of squid migration reflects North Atlantic climate variability. Proceedings of the Royal Society of London Series B-Biological Sciences 268(1485):2607-2611.

Simmonds, M. P. and L. F. Lopez-Jurado. 1991. Whales and the military. Nature 351:448.

Sinclair, E. H. 1992. Stomach contents of four short-finned pilot whales (*Globicephala macrorhynchus*) from the Southern California Bight. Marine Mammal Science 8:76-81.

Skals, N., D. Plepys, and C. Lofstedt. 2003. Foraging and mate-finding in the silver Y moth, *Autographa gamma* (Lepidoptera: Noctuidae) under the risk of predation. Oikos 102:351-357.

Skalski, J. R., W. H. Pearson and C. I. Malme 1992. Effects of sounds from a geophysical survey device on catch-per-unit-effort in a hook-and-line fishery for rockfish (*Sebastes* spp.). Canadian Journal of Fisheries and Aquatic Sciences 49:1357-1365.

Smith, M. E., A. S. Kane, and A. N. Popper. 2004. Acoustical stress and hearing sensitivity in fishes: does the linear threshold shift hypothesis hold water? The journal of Experimental Biology 207:3591 - 3602.

Smith, M. E., L. E. Wysocki, and A. N. Popper. 2006. Effects of background sound on fish. The Journal of the Acoustical Society of America 119:1.

Smith, S. C. and H. Whitehead. 2000. The diet of Galapagos sperm whales *Physeter macrocephalus* as indicated by fecal sample analysis. Marine Mammal Science 16:11.

Smith, T.I.J. 1985. The Fishery, Biology, and Management of Atlantic sturgeon, *Acipenser oxyrhynchus*, in North America. Environmental Biology of Fishes 14(1): 61-72.

Smith, T.I.J. and J.P. Clugston. 1997. Status and Management of Atlantic Sturgeon, *Acipenser oxyrinchus*, in North America. Environmental Biology of Fishes 48: 335-346.

Smultea, M., Mobley, J.R., Fertl, D. and Fulling, G.L. 2008. An unusual reaction and other observations of sperm whales near fixed-wing aircraft. Gulf and Caribbean Research 20: 75-80.

Solomon, S., G-K. Plattner, R. Knutti and P. Friedlingstein. 2009. Irreversible climate change due to carbon dioxide emissions. Proceedings of the National Academy of Sciences. Vol 106(6): 1704-1709. www.pnas.org/cgi/doi/10.1073/pnas.0812721106

Southall, B.L., and co-authors. 2007. Marine Mammal Noise Exposure Criteria: Initial Scientific Recommendations. Aquatic Mammals, Volume 33, Number 4, 2007.

Southall, B. L., R. J. Schusterman, and D. Kastak. 2000. Masking in three pinnipeds: underwater, low-frequency critical ratios. The Journal of the Acoustical Society of America 108:1322.

Spitz, S. S., L. M. Herman, A. A. Pack, and M. H. Deakos. 2002. The relation of body size of male humpback whales to their social roles on the Hawaiian winter grounds. The Canadian Journal of Zoology 80:1938-1947.

Squiers, T.S., M. Smith. 1979. Distribution and abundance of shortnose and Atlantic sturgeon in the Kennebec River Estuary. Report. Compl. Rep. Maine Mar. Resour. 53 pages.

Stafford, K. 2000. Blue whale calls recorded in the Gulf of Alaska. The Journal of the Acoustical Society of America 108:2614.

Stafford, K. M., C. G. Fox, and D. S. Clark. 1998. Long-range acoustic detection, localization of blue whale calls in the northeast Pacific Ocean. The Journal of the Acoustical Society of America 104:3616.

Stafford, K. M., S. L. Nieukirk, and C. G. Fox. 1999. An acoustic link between blue whales in the eastern tropical Pacific and the Northeast Pacific. Marine Mammal Science 15:11.

Stafford, K. M., S. L. Nieukirk, and C. G. Fox. 1999. Low-frequency whale sounds recorded on hydrophones moored in the eastern tropical Pacific. The Journal of the Acoustical Society of America 106:3687.

Stafford, K. M., S. L. Nieukirk, and C. G. Fox. 2001. Geographic and seasonal variation of blue whale calls in the North Pacific. Journal of Cetacean Research and Management 3.

Steig, T. W. and S. V. Johnston. 2000. The development of a system for remotely monitoring the three-dimensional movement of acoustically tagged fish. The Journal of the Acoustical Society of America 108:1.

Stein, A. B., K. D. Friedland, and M. Sutherland. 2004. Atlantic sturgeon marine distribution and habitat use along the northeastern coast of the United States. Transactions of the American Fisheries Society 133:527-537.

Stevenson, J. T. and D. H. Secor. 1999. Age determination and growth of Hudson River Atlantic sturgeon, Acipenser oxyrinchus. Fishery Bulletin 97:153-166.

Stevick, P. T. 1999. Age-length relationships in humpback whales: A comparison of strandings in the western North Atlantic with commercial catches. Marine Mammal Science 15:13.

Stevick, P. T., J. Allen, M. Berube, P. J. Clapham, S. K. Katona, F. Larsen, J. Lien, D. K. Mattila, P. J. Palsboll, J. Robbins, J. Sigurjonsson, T. D. Smith, N. Oien, and P. S. Hammond. 2003. Segregation of migration by feeding ground origin in North Atlantic humpback whales (*Megaptera novaeangliae*). Journal of Zoology 259:231-237.

Stocker, M. 2002. Fish, mollusks and other sea animals' use of sound, and the impact of anthropogenic noise in the marine acoustic environment. The Journal of the Acoustical Society of America 112:2431.

Stocker, M. 2002. Sounding the alarm on underwater noise. Nature 420:740.

Sugahara, K., H. Shimogori, T. Okuda, T. Takemoto, M. Hashimoto, and H. Yamashita. 2004. Cochlear administration of adenosine triphosphate facilitates recovery from acoustic trauma (temporary threshold shift). Journal for Oto-Rhino-Laryngology and its Borderlands 66:80-84.

Supin, A. Y., P. E. Nachtigall, and W. W. L. Au. 2002. Evoked-potential study of echolocation in whales and dolphins. The Journal of the Acoustical Society of America 112:2313.

Sutherland, W. J. 1996. From individual behavior to population ecology. Oxford University Press, Oxford, United Kingdom.

Swartzman, G. L., D. V. Holliday, K. O. Coyle, and J. M. Napp. 2000. Comparison and evaluation of methods for distinguishing among several functional classes of zooplankton and fish using acoustic backscatter data at four frequencies. The Journal of the Acoustical Society of America 108:2490.

Swingle, W. M., S. G. Barco, T. D. Pitchford, W. A. McLellan, and D. A. Pabst. 1993. Appearance of juvenile humpback whales feeding in the nearshore waters of Virginia. Marine Mammal Science 9:309-315.

Szymanski, M. D., D. E. Bain, K. Kiehl, S. Pennington, S. Wong, and K. R. Henry. 1999. Killer whale (*Orcinus orca*) hearing: Auditory brainstem response and behavioral audiograms. The Journal of the Acoustical Society of America 106:1134.

Talpalar, A. and Y. Grossman. 2005. Sonar versus whales: Noise may disrupt neural activity in deep-diving cetaceans. Undersea & Hyperbaric Medicine 32:135.

Teloni, V. 2005. Patterns of sound production in diving sperm whales in the northwestern Mediterranean. Marine Mammal Science 21:12.

Teloni, V., W. M. X. Zimmer, and P. L. Tyack. 2005. Sperm whale trumpet sounds. Bioacoustics 15:12.

Thewissen, J. G. M. and E. M. Williams. 2002. The early radiations of Cetacea (Mammalia): Evolutionary pattern and developmental correlations. Annual Review of Ecology and Systematics 33:73-90.

Thode, A. 2004. Tracking sperm whale (*Physeter macrocephalus*) dive profiles using a towed passive acoustic array. The Journal of the Acoustical Society of America 116:9.

Thode, A. 2005. Three-dimensional passive acoustic tracking of sperm whales (*Physeter macrocephalus*) in ray-refracting environments. The Journal of the Acoustical Society of America 118:10.

Thode, A., D. K. Mellinger, S. Stienessen, A. Martinez, and K. Mullin. 2002. Depth-dependent acoustic features of diving sperm whales (*Physeter macrocephalus*) in the Gulf of Mexico. The Journal of the Acoustical Society of America 112:14.

Thode, A. M., G. L. D'Spain, and W. A. Kuperman. 2000. Matched-field processing, geoacoustic inversion, and source signature recovery of blue whale vocalizations. The Journal of the Acoustical Society of America 107:1286.

Thode, A. M., D. K. Mellinger, S. Stienessen, A. Martinez, and K. Mullin. 2001. Analysis of frequency structure of sperm whale clicks as a function of dive depth and animal orientation. The Journal of the Acoustical Society of America 109:2390.

Thompson, P. O., L. T. Findley, O. Vidal, and W. C. Cummings. 1996. Underwater sounds of blue whales, *Balaenoptera musculus*, in the Gulf of California, Mexico. Marine Mammal Science 12:5.

Tiemann, C., A. Thode, J. Straley, K. Folkert, and V. O'Connell. 2005. Model-based passive acoustic tracking of sperm whale foraging behavior in the Gulf of Alaska. The Journal of the Acoustical Society of America 118:1909.

Todd, S., P. T. Stevick, J. Lien, F. Marques, and D. Ketten. 1996. Behavioral effects of exposure to underwater explosions in humpback whales (*Megaptera novaeangliae*). The Canadian Journal of Zoology 74:1661-1672.

Trevorrow, M. V. 1996. Multifrequency acoustic investigations of juvenile and adult fish in Lake Biwa, Japan. The Journal of the Acoustical Society of America 100:11.

Turnpenny, A.W.H. and J.R. Nedwell. 1994. The effects on marine fish, diving mammals and birds of underwater sound generated by seismic surveys. Fawley Aquatic Research laboratories Ltd. FCR 089/94. October 1994. 40 pp.

Tyack, P. L. 2000. Functional aspects of cetacean communication. Pages 270-307 *in* J. Mann, R. C. Connor, P. L. Tyack, and H. Whitehead, editors. Cetacean societies. Field studies of dolphins and whales. University of Chicago Press, Chicago, Illinois.

Tyack, P. L., R. C. Connor, J. Mann, and H. Whitehead. 2000. Epilogue: the future of behavioral research on cetaceans in the wild. Pages 333-339 *in* J. Mann, R. C. Connor, P. L. Tyack, and H. Whitehead, editors. Cetacean societies. Field studies of dolphins and whales. University of Chicago Press, Chicago, Illinois.

U.S. Dept. of Commerce, National Oceanic and Atmospheric Administration. 2006. Fact sheet: Small diesel spills (500-5,000 gallons). NOAA Scientific Support Team, Hazardous Materials Response and Assessment Division, National Oceanic and Atmospheric Administration, Seattle, WA. 2 pp. Available at: http://response.restoration.noaa.gov/book_shelf/974_diesel.pdf.

U.S. Dept. of the Interior, Minerals Management Service (MMS). 1998. North Carolina/Minerals Management Service Technical Workshop on Manteo Unit Exploration. February 4-5, 1998. Minerals Management Service, Gulf of Mexico Region, New Orleans, LA. OCS Report MMS 98-0024. Available at: http://www.gomr.boemre.gov/PI/PDFImages/ESPIS/3/3225.pdf

U.S. Dept. of the Interior, (MMS). 2008. Gulf of Mexico OCS Oil and Gas Lease Sales: 2009-2012; Central Planning Area Sales 208, 213, 216, and 222; Western Planning area Sales 210, 215, and 218 – Final Supplemental Environmental Impact Statement. U.S. Dept. of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, LA. OCS EIS/EA MMS 2008-041.

U.S. Department of the Interior, BOEM. 2012a. Atlantic OCS Proposed Geological and Geophysical Activities Mid-Atlantic and South Atlantic Planning Areas Draft Programmatic Environmental Impact Statement, U.S. Dept. of the Interior, Minerals Management Service, Gulf of Mexico OCS Region, New Orleans, LA, OCS EIS/EA BOEM 2012-005. Available at: http://www.boem.gov/BOEM-Newsroom/Library/Publications/2012/BOEM-2012-005-vol1-pdf.aspx

U.S. Department of the Interior, BOEM. 2012b. Biological Assessment for Atlantic OCS Proposed Geological and Geophysical Activities Mid-Atlantic and South Atlantic Planning Areas, May 22, 2012. 225 pp.

U.S. Dept. of the Interior, BOEM and BSEE. 2012a. Joint NTL 2012-G02. Notice to Lessees and Operators (NTL) of Federal oil, gas, and sulphur leases and pipeline right of-way holders in the OCS, Gulf of Mexico OCS Region. Implementation of Seismic Survey Mitigation Measures and Protected Species Observer Program. Available at: http://www.bsee.gov/Regulations-and-Guidance/Notices-to-Lessees/2012/2012-JOINT-G02-pdf.aspx.

U.S. Dept. of the Interior, Bureau of Ocean Energy Management (BOEM) and Bureau of Safety and Environmental Enforcement (BSEE). 2012b. Joint NTL 2012-G01. Notice to Lessees and Operators (NTL) of Federal oil, gas, and sulphur leases and pipeline right of-way holders in the OCS, Gulf of Mexico OCS Region. Vessel Strike Avoidance and Injured/Dead Protected Species Reporting. Available at: http://www.boem.gov/Regulations/Notices-To-Lessees/2012/2012-JOINT-G01-pdf.aspx.

U.S. Dept. of the Interior, BSEE. 2012. BSEE NTL 2012-G01. Notice to Lessees and Operators (NTL) of Federal oil, gas, and sulphur leases and pipeline right of-way holders in the OCS, Gulf of Mexico OCS Region. Marine trash and debris awareness and elimination. Available at: http://www.bsee.gov/Regulations-and-Guidance/Notices-to-Lessees/2012/2012-BSEE-G01-pdf.aspx

U.S. Marine Mammal Commission. 2003. Annual Report to Congress 2002. Marine Mammal Commission, Bethesda, Maryland.

U.S. Global Change Research Program. 2009—Global Climate Change Impacts in the United States. Thomas R. Karl, Jerry M. Melillo, and Thomas C. Peterson, (eds.). Cambridge University Press, 2009. Available at: www.globalchange.gov/usimpacts.

U.S. Department of Interior (DOI), BOEM and BSEE. 2012b. Joint BOEM-BSEE NTL 2012-G01 (*Vessel Strike Avoidance and Injured/Dead Protected Species Reporting.* Available at: http://www.bsee.gov/Regulations-and-Guidance/Notices-to-Lessees-and-Operators.aspx

U.S. Department of Transportation. 2011. Vessel Calls Snapshot, 2010. U.S. Department of Transportation, Maritime Administration, May 2011. http://www.marad.dot.gov/).

U.S. Department of Transportation. 2010. Atlantic Coast U.S. Seaports Fact Sheet. U.S. Department of Transportation, Research and Innovative Technology Administration. October 2010. Available at: www.bts.gov

U.S. Department of Transportation, Maritime Administration. 2009. America's Ports and Intermodal Transportation System. Available at: www.marad.dot.gov

Urban, J., V. F. d. Sahagun, M. L. Jones, S. L. Swartz, B. Mate, A. Gomez-Gallardo, and M. Guerrero-Ruiz. 2004. Gray whales with loss of flukes adapt and survive. Marine Mammal Science 20:4.

Urban-R, J., S. Ramirez-S, and J. C. Salinas-V. 1994. First record of bottlenose whales, Hyperoodon sp , in the Gulf of California. Marine Mammal Science 10:471.

Van Eenennaam, J.P., S.I. Doroshov, G.P. Moberg, J.G. Watson, D.S. Moore, and J. Linares. 1996. Reproductive conditions of the Atlantic sturgeon (*Acipenser oxyrinchus*) in the Hudson River. Estuaries 19(4):769-777.

Van Parijs, S. M. and P. J. Corkeron. 2001. Boat traffic affects the acoustic behaviour of Pacific humpback dolphins, *Sousa chinensis*. Journal of the Marine Biological Association of the UK 81:533-538.

Van Waerebeek K and R. Leaper. 2008. Second Report of the IWC Vessel Strike Data Standardisation Working Group. *Paper SC/60/BC5 presented to the IWC Scientific Committee, May 2008 (unpublished). 8pp.*

Van Waerebeek, K., M.F. Van Bressem, F. Felix, J. Alfaro-Shigueto, A. Garcia-Godos, L. Chavez-Lisambart, K. Onton, D. Montes, and R. Bello. 1997. Mortality of dolphins and porpoises in coastal fisheries off Peru and southern Ecuador in 1994. Biological Conservation 81:43-49.

Vladykov, V. D. and J. R. Greely. 1963. Order Acipenseroidei. In: Fishes of Western North Atlantic. Sears Foundation. Marine Research, Yale Univ. 1 630 pp.

von Gierke, H. E. and W. D. Ward. 1998. Chapter 26. Criteria for noise and vibration exposure. Pages 26.21 - 26.17 *in* C. M. Harris, editor. Handbook of acoustical measurements and noise control. Acoustical Society of America, Woodbury, New York.

Wade, P. R. and T. Gerrodette. 1993. Estimates of cetacean abundance and distribution in the eastern tropical Pacific. Rep. Int. Whal. Commn. 43:477-493.

Wahlberg, M. 2002. The acoustic behaviour of diving sperm whales observed with a hydrophone array. Journal of Experimental Marine Biology and Ecology 281:53-62.

Wahlberg, M., A. Frantzis, P. Alexiadou, P. T. Madsen, and B. Mo-hl. 2005. Click production during breathing in a sperm whale (*Physeter macrocephalus*). The Journal of the Acoustical Society of America 118:3404-3407.

Walker, R. J., E. O. Keith, A. E. Yankovsky, and D. K. Odell. 2005. Environmental correlates of cetacean mass stranding sites in Florida. Marine Mammal Science 21:9.

Walker, W. A. and M. B. Hanson. 1999. Biological observations on Stejneger's beaked whale, *Mesoplodon stejnegeri*, from strandings on Adak Island, Alaska. Marine Mammal Science 15:15.

Walker, W. A., J. G. Mead, and R. L. Brownell Jr. 2002. Diets of Baird's beaked whales, *Berardius hairdii*, in the southern Sea of Okhotsk and off the Pacific coast of Honshu, Japan. Marine Mammal Science 18:18.

Wardle, C.S., T.J. Carter, G.G. Urquhart, A.D.F. Johnstone, A.M. Ziolkowski, G. Hampson, and D. Mackie. 2001. Effects of seismic airguns on marine fish. Continental Shelf Research 21:1005-1027.

Waring, G. T., T. Hamazaki, D. Sheehan, G. Wood, and S. Baker. 2001. Characterization of beaked whale (*Ziphiidae*) and sperm whale (*Physeter macrocephalus*) summer habitat in shelf-edge and deeper waters off the northeast U.S. Marine Mammal Science 17:703.

Waring, G.T., E. Josephson, C.P. Fairfield-Walsh, K. Maze-Foley. 2007. U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments—2007. NOAA Tech. Memo. NMFS-NE-205. 410 p.

Waring, G.T., E. Josephson, C.P. Fairfield-Walsh, K. Maze-Foley. 2009. U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments—2008. NOAA Tech. Memo. NMFS-NE-210. 440 p

Waring GT, Josephson E, Maze-Foley K, Rosel, PE, editors. 2011. U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments -- 2010. NOAA Tech Memo NMFS NE 219; 595 p. Available from: National Marine Fisheries Service, 166 Water Street, Woods Hole, MA 02543-1026, or online athttp://www.nefsc.noaa.gov/nefsc/publications/

Waring, G.T., E. Josephson, K. Maze-Foley, and P. E. Rosel. 2012. U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments—2011. NOAA Tech. Memo. NMFS-NE-221. 319 p.

Walther, G. 2010. Community and ecosystem responses to recent climate change. Philosophical Transaction of the Royal Society of London, B. 365: 2019-2024.

Watkins, W. A. 1986. Whale reactions to human activities in Cape Cod waters. Marine Mammal Science 2:251-262.

Watkins, W. A., M. A. Daher, N. A. DiMarzio, A. Samuels, D. Wartzok, K. M. Fristrup, D. P. Gannon, P. W. Howey, and R. R. Maiefski. 1999. Sperm whale surface activity from tracking by radio and satellite tags. Marine Mammal Science 15:23.

Watkins, W. A., M. A. Daher, N. A. DiMarzio, A. Samuels, D. Wartzok, K. M. Fristrup, P. W. Howey, and R. R. Maiefski. 2002. Sperm whale dives tracked by radio tag telemetry. Marine Mammal Science 18:14.

Watkins, W. A., M. A. Daher, K. M. Fristrup, T. J. Howald, and G. N. di Sciara. 1993. Sperm whales tagged with transponders and tracked underwater by sonar. Marine Mammal Science 9:55-67.

Watkins, W. A., K. E. Moore, and P. L. Tyack. 1987. Sperm whale acoustic behaviors in the southeast Caribbean. Cetology 49:1-15.

Watkins, W. A., J. Sigurjonsson, D. Wartzok, R. R. Maiefski, P. W. Howey, and M. A. Daher. 1996. Fin whale tracked by satellite off Iceland. Marine Mammal Science 12:5.

Watkins, W. A. and P. Tyack. 1991. Reaction of sperm whales (*Physeter catadon*) to tagging with implanted sonar transponder and radio tags. Marine Mammal Science 7:409-412.

Watwood, S. L., P. J. O. Miller, M. Johnson, P. T. Madsen, and P. L. Tyack. 2006. Deep-diving foraging behaviour of sperm whales (*Physeter macrocephalus*). The Journal of Animal Ecology 75:814-825.

Weilgart, L. and H. Whitehead. 1997. Group-specific dialects and geographical variation in coda repertoire in South Pacific sperm whales. Behavioral Ecology and Sociobiology 40:277-285.

Weller, D. W., B. Wursig, H. Whitehead, J. C. Norris, S. K. Lynn, R. W. Davis, N. Clauss, and P. Brown. 1996. Observations of an interaction between sperm whales and short-finned pilot whales in the Gulf of Mexico. Marine Mammal Science 12:6.

Weston, D. E. 1992. Mechanisms of ocean acoustic attenuation: Scattering by internal solitons, by sea surface waves, and by fish. The Journal of the Acoustical Society of America 92:3435.

Whitehead, H. 1999. Variation in the visually observable behavior of groups of Galapagos sperm whales. Marine Mammal Science 15:17.

Whitehead, H. 2000. Density-dependent habitat selection and the modeling of sperm whale (*Physeter macrocephalus*) exploitation. Canadian Journal of Fisheries and Aquatic Sciences 57:223-230.

Whitehead, H. 2001. Analysis of animal movement using opportunistic individual identifications: Application to sperm whales. Ecology 82:1417-1432.

Whitehead, H. 2001. Direct estimation of within-group heterogeneity in photo-identification of sperm whales. Marine Mammal Science 17:11.

Whitehead, H. 2003. Sperm whales. University of Chicago Press, Chicago, Illinois.

Whitehead, H., J. Christal, and S. Dufault. 1997. Past and distant whaling and the rapid decline of sperm whales off the Galapagos Islands. Conservation Biology 11:1387-1396.

Whitehead, H., J. Christal, and P. L. Tyack. 2000. Studying cetacean social structure in space and time: innovative techniques. Pages 65-87 *in* J. Mann, R. C. Connor, P. L. Tyack, and H. Whitehead, editors. Cetacean societies. Field studies of dolphins and whales. University of Chicago Press, Chicago, Illinois.

Whitehead, H., J. Gordon, E. A. Mathews, and K. R. Richard. 1990. Obtaining skin samples from living sperm whales. Marine Mammal Science 6:316-326.

Whitehead, H., S. Gowans, A. Faucher, and S. W. McCarrey. 1997. Population analysis of northern bottlenose whales in the Gully, Nova Scotia. Marine Mammal Science 13:13.

Whitehead, H. and J. Mann. 2000. Female reproductive strategies of cetaceans: life histories and calf care. Pages 219-246 *in* J. Mann, R. C. Connor, P. L. Tyack, and H. Whitehead, editors. Cetacean societies. Field studies of dolphins and whales. University of Chicago Press, Chicago, Illinois.

Whitehead, H., R. R. Reeves, and P. L. Tyack. 2000. Science and the conservation, protection, and management of wild cetaceans. Pages 308-332 *in* J. Mann, R. C. Connor, P. L. Tyack, and H. Whitehead, editors. Cetacean societies. Field studies of dolphins and whales. University of Chicago Press, Chicago, Illinois.

Whitehead, H. and L. Weilcart. 2000. The sperm whale: social females and roving males. Pages 154-172 *in* J. Mann, R. C. Connor, P. L. Tyack, and H. Whitehead, editors. Cetacean societies. Field studies of dolphins and whales. University of Chicago Press, Chicago, Illinois.

Wiedenmann, J., K.A. Cresswell, J. Goldbogen, J. Potvin, M. Mangel. 2011. Exploring the effects of reductions in krill biomass in the Southern Ocean on blue whales using a state-dependent foraging model. Ecological Modelling 222(18), 3366-3379.

Weilgart, L.S. 2007. A brief review of known effects of noise on marine mammals. International Journal of Compparative Psychology, 20, 159-168.

Wenzel, F. W., D. K. Mattila and P. J. Clapham 1988. *Balaenoptera musculus* in the Gulf of Maine. Marine Mammal Science 4(2):172-175.

Wiley, D.N., M. Thompson, R.M. pace III, and J. Levenson. 2011). Modeling speed restrictions to mitigate lethal collisions between ships and whales in the Stellwagen Bank National Marine Sanctuary, USA. Biological Conservation 144(2011) 2377-2381.

Williams, R. M., A. W. Trites, and D. E. Bain. 2002. Behavioral responses of killer whales (*Orcinus orca*) to whale-watching boats: Opportunistic observations and experimental approaches. Journal of Zoology 256:255-270.

Williams, T. M., R. W. Davis, L. A. Fuiman, J. Francis, B. J. Le Boeuf, M. Horning, J. Calambokidis, and D. A. Croll. 2000. Sink or swim: strategies for cost-efficient diving by marine mammals. Science 288:133-136.

Wysocki, L. E., J. P. Dittami, and F. Ladich. 2006. Ship noise and cortisol secretion in European freshwater fishes. Biological Conservation 128:501-508.

Würsig, B., S. K. Lynn, T. A. Jefferson, and K. D. Mullin. 1998. Behavior of cetaceans in the northern Gulf of Mexico relative to survey ships and aircraft. Aquatic Mammals 24:41-50.

Ye, Z. 1995. Theoretical description of possible detection of swimbladdered fish in forward scatter. The Journal of the Acoustical Society of America 98:9.

Ye, Z. 1996. Acoustic scattering from fish swimbladders. The Journal of the Acoustical Society of America 99:8.

Ye, Z. 1996. On acoustic attenuation by swimbladder fish. The Journal of the Acoustical Society of America 100:4.

Ye, Z. 1997. Erratum: "On acoustic attenuation by swimbladder fish" (J Acoust Soc Am 100, 669-672 (1996)). The Journal of the Acoustical Society of America 101:1.

Ye, Z. and D. Chu. 2003. Comment on "Multiple scattering in a reflecting cavity: Application to fish counting in a tank" (J Acoust Soc Am 109, 2587-2597 (2001)) (L). The Journal of the Acoustical Society of America 113:2.

Ye, Z. and D. M. Farmer. 1994. Acoustic scattering from swim-bladder fish at low frequencies. The Journal of the Acoustical Society of America 96:951.

Yochem, P. K. and S. Leatherwood 1985. Blue whale *Balaenoptera musculus* (Linnaeus, 1758). In: Ridgway SH, Harrison R, editors. Handbook of Marine Mammals, vol. 3: The Sirenians and Baleen Whales.:London: Academic Press. p 193-240.

Yoshida, H. and H. Kato. 1999. Phylogenetic relationships of Bryde's whales in the western North Pacific and adjacent waters inferred from mitochondrial DNA sequences. Marine Mammal Science 15:18.

Yurk, H., L. Barrett-Lennard, J. K. B. Ford, and C. O. Matkin. 2002. Cultural transmission within maternal lineages: Vocal clans in resident killer whales in southern Alaska. The Canadian Journal of Zoology 63:1103-1119.

Zimmer, W. M. X., M. P. Johnson, A. D'Amico, and P. L. Tyack. 2003. Combining data from a multisensor tag and passive sonar to determine the diving behavior of a sperm whale (*Physeter macrocephalus*). IEEE Journal of Oceanic Engineering 28:16.

Zimmer, W. M. X., P. L. Tyack, M. P. Johnson, and P. T. Madsen. 2005. Three-dimensional beam pattern of regular sperm whale clicks confirms bent-horn hypothesis. The Journal of the Acoustical Society of America 117:13.

Zykov, M., and J. MacDonnell. 2013. Sound source characterizations for the collaborative baseline survey offshore Massachusetts Final report: Side-scan sonar, sub-bottom profiler, and the R/V Small Research Vessel experimental. JASCO.