NATIONAL MARINE FISHERIES SERVICE
ENDANGERED SPECIES ACT - SECTION 7 CONSULTATION

BIOLOGICAL OPINION

Agencies: United States Navy
National Marine Fisheries Service

Activity: Proposed Employment of Surveillance Towed Array Sensor System Low Frequency Active Sonar

Proposed regulations to authorize the Navy to take marine mammals incidental to its employment of Surveillance Towed Array Sensor System Low Frequency Active Sonar

Consultation Conducted By: National Marine Fisheries Service

Date Issued: May 30, 2002

Approved by: David

This document constitutes the National Marine Fisheries Service's (NMFS) biological opinion on the U.S. Navy's proposed use of Surveillance Towed Array Sensor System Low Frequency Active Sonar on threatened and endangered species, designated critical habitat, and species that have been proposed for listing. This biological opinion has been prepared in accordance with section 7 of the Endangered Species Act (ESA) of 1973, as amended (16 U.S.C. 1531 et seq.). A complete administrative record of this consultation is on file with NMFS' Office of Protected Resources, Endangered Species Division, Silver Spring, Maryland [Consultation No. F/FPR/2000/00483].

This Biological Opinion is based on information provided in the January 2001 Final Overseas Environmental Impact Statement and Environmental Impact Statement for Surveillance Towed Array Sensor System Low Frequency Active (SURTASS LFA) Sonar (Navy 2001), the October 1999 Biological Assessment for the Employment of the Surveillance Towed Array Sensor System Low Frequency Active (SURTASS LFA) Sonar (Navy 1999); a February 1999 report on Marine Vertebrates and Low Frequency Sound: Technical Report for LFA EIS prepared by the Marine Mammal and Seabird Ecology Group of the University of California, Santa Cruz, Institute of Marine Sciences (Croll *et al.* 1999); various documents NMFS' Marine Mammal Conservation Division prepared in support of proposed and final regulatory amendments that would authorize the U.S. Navy to take small numbers of marine mammals incidental to the employment of SURTASS LFA; and other published and unpublished scientific and technical information.

CONSULTATION HISTORY

On May 18, 1998, the U.S. Navy sent a letter to NMFS requesting a list of the threatened, endangered, proposed, and candidate species and designated and proposed critical habitat. On January 27, 1999, NMFS responded to this request by providing the Navy with a list of species and designated critical habitat. That letter highlighted the potential effects of the proposed action on threatened and endangered large whales, Steller sea lions, and sea turtles. That letter did not include any species of salmon in the list of species that could be potentially affected by the proposed SURTASS LFA sonar.

On August 12, 1999, the Navy submitted an application for a small take authorization under section 101(a)(5)(A) of the Marine Mammal Protection Act for the taking of marine mammals incidental to operation of the SURTASS LFA sonar for a period of time not to exceed 5 years, beginning in Fiscal Year 2000.

The Navy submitted a request for formal consultation pursuant to section 7 of the ESA on October 4, 1999. On October 22, 1999, NMFS published an Advance Notice of Proposed Rulemaking on the U.S. Navy application and invited interested persons to submit comments, information, and suggestions concerning the application and the structure and content of regulations (64 FR 57026).

On January 22, 2001, representatives of NMFS met with the U.S. Navy to discuss their proposed SURTASS LFA project and their desire to complete section 7 consultation on the project. The Navy proposed to begin using the SURTASS LFA sonar by June of 2002, depending upon completion of the NEPA process and issuance of a Record of Decision supporting deployment. Attendees agreed to group the Navy's proposal with NMFS' proposal to modify its existing regulations to authorize the Navy to take marine mammals incidental to its employment of SURTASS LFA sonar in a single biological opinion rather than prepare separate biological opinions on both actions.

On January 30, 2001, representatives of NMFS met with the U.S. Navy to discuss the schedule and procedures for the section 7 consultation on the proposed SURTASS LFA project. The group also discussed the action area that would be included in the section 7 consultation.

On March 19, 2001, NMFS published a proposed rule to authorize the U.S. Navy to take small numbers of marine mammals incidental to operation of SURTASS LFA sonar and requested comments, information, and suggestions concerning the request and the regulations proposed to govern the take (66 FR 15375).

On April 16, 2001, NMFS published a notice of public hearings and extended the deadline for public comments. On May 15, 2001, NMFS extended the deadline for public comments a second time. Between April and May 2001, representatives of NMFS held public hearings on a proposed small take authorization in Los Angeles, California, Honolulu, Hawaii, and Silver Spring, Maryland.

DESCRIPTION OF THE PROPOSED ACTIONS

This biological opinion considers two interrelated actions proposed by the U.S. Navy and the National Marine Fisheries Service's Marine Mammal Conservation Division. The United States Navy (Navy) proposes to employ the Surveillance Towed Array Sensor System Low Frequency Active (SURTASS LFA) sonar on 4 vessels during routine training and testing as well as during military operations. In an interrelated action, the National Marine Fisheries Service's Marine Mammal Conservation Division proposes to amend its regulations governing the take and importation of marine mammals (50 CFR Part 216) to make it possible for NMFS to issue annual letters of authorization that would allow the U.S. Navy to take marine mammals incidental to the U.S. Navy's operations of SURTASS LFA sonar for a five-year period ending in 2007.

The purpose of the Navy's proposed action is to meet the United States' need for an improved ability to detect quieter and harder-to-find foreign submarines at long range to provide U.S. forces with adequate time to respond to potential submarine threats.

The SURTASS LFA sonar[1] system is a long-range, low frequency sonar (between 100 and 500 Hertz(Hz)) that has both active and passive components (See Figure 1). The active component of the system (LFA) consists of up to 18 low-frequency acoustic-transmitting source elements (called projectors) that are suspended from a cable beneath a ship. The projectors transform electrical energy to mechanical energy by setting up vibrations, or pressure disturbances, with the water to produce the active sound (which is called a "pulse" or a "ping"). SURTASS LFA's transmitted beam is omnidirectional (full 360 degrees) in the horizontal. The nominal water depth of the center of the array is 400 ft (122 m), with a narrow vertical beamwidth that can be steered above or below the horizontal. The source level of an individual projector in the SURTASS LFA sonar array is approximately 215 dB, and the sound field of the array can never have a sound pressure level higher than that of an individual projector. The nominal, minimum water depth that the SURTASS LFA vessel will operate is 200 m (656.2 ft); the shallowest water depth is 100 m (328.1 ft).

The typical SURTASS LFA sonar signal is not a constant tone, but is a transmission of various signal types that vary in frequency and duration (including continuous wave and frequency-modulated signals). The Navy refers to a complete sequence of sound transmissions as a "ping" which can range from between 6 and 100 seconds, with no more than 10 seconds at any single frequency. The time between pings will typically range from 6 to 15 minutes. The Navy can control the average duty cycle (the ratio of sound "on" time to total time) for the system but the duty cycle cannot be greater than 20 percent; the Navy anticipates a typical duty cycle between 10 and 15 percent.

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[1] Sonar is an acronym for **so**und **na**vigation and **r**anging, and its definition includes any system that uses underwater sound, or acoustics, for observations and communications.

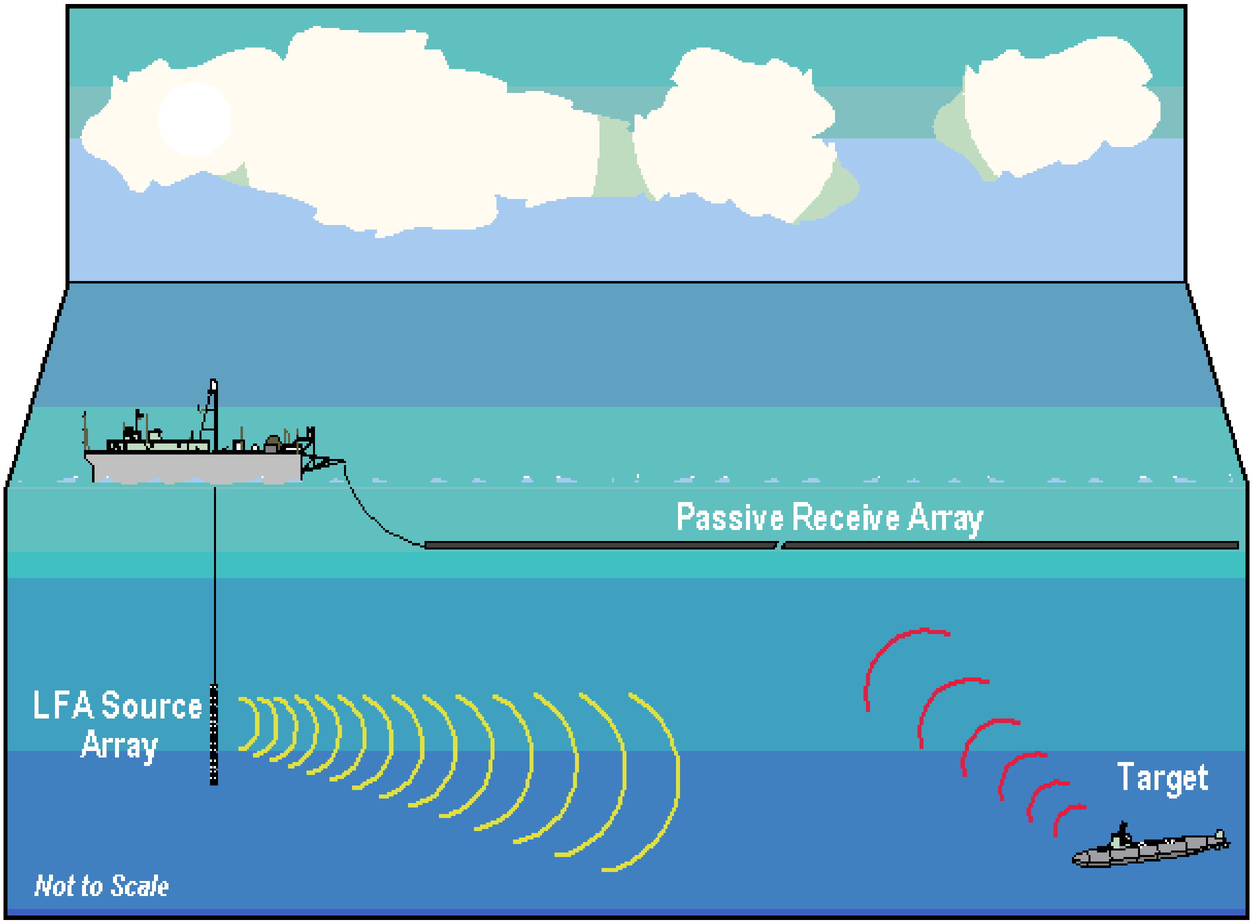

Figure 1. Schematic of the SURTASS LFA sonar system

The passive or listening component of the system (SURTASS) uses hydrophones to detect echoes of the signal returning from submerged objects, such as submarines. The hydrophones are mounted on a horizontal array that is towed behind the ship. The SURTASS LFA sonar ship maintains a minimum speed of 3.0 knots (5.6 km/hr; 3.4 mi/hr) in order to keep the array properly deployed. The return signals, which are usually below background or ambient noise levels, are then processed and evaluated to identify and classify potential underwater threats.

The SURTASS LFA sonar vessel would operate independently of, or in conjunction with, other Naval air, surface or submarine assets. The vessel would generally travel in straight lines or racetrack patterns depending on the operational scenario. A nominal SURTASS LFA sonar deployment schedule for a single vessel would consist of about 270 days/year at sea. Nominal at-sea missions would consist of two 9-day exercise segments that would occur over a 30-day period. Active sonar operations could be conducted up to 20 hrs during an exercise day, although the system would actually transmit for a maximum of about 4 hrs/day (resulting in 432 hrs of active transmission time per year for each SURTASS LFA sonar system in operation based on a maximum duty cycle of 20 percent). The remaining 12 days of the missions would consist of transiting or repositioning the vessel. In a nominal year there could be a maximum of 9 missions, six of which would involve the employment of SURTASS LFA sonar in the active mode and three of which would employ the system only in the passive mode (that is, use of the SURTASS component). Between missions, ships would spend a total of about 95 days in port for maintenance and repair.

In their Draft and Final EIS for this project, the Navy proposed to employ SURTASS LFA on two vessels in the Pacific and Indian Oceans and two vessels in the Atlantic Ocean and Mediterranean Sea. However, only one SURTASS LFA sonar system is currently available for deployment with a second system expected to become available shortly. Deliveries of the third and fourth systems have been postponed until after FY 2007. Because of this delay and the 5-year duration of NMFS' proposed small-take authorization, NMFS is authorizing marine mammal harassment takings for only 2 SURTASS LFA sonar systems. With two vessels, there could be up to 6 SURTASS LFA sonar missions or equivalents per vessel each year for five years.

At present, the R/V *Cory Chouest* is the only vessel equipped with SURTASS LFA sonar. The Navy intends to employ this ship in the Pacific Ocean, but may employ it in other parts of the world. However, the Navy will not operate SURTASS LFA sonar in polar waters (Navy 2001).

Mitigation Measures

Based on the SURTASS LFA Sonar Final EIS, the Navy proposes to use an active monitoring program and geographic restrictions to avoid potential, adverse effects of SURTASS LFA sonar on marine mammals. In its rule making under the Marine Mammal Protection Act (MMPA), NMFS has also required the Navy to implement additional interim operational procedures. The following mitigation includes both.

Monitoring

To avoid potential injuries to marine mammals (and possibly sea turtles), the Navy proposes to detect animals within an area they call the "LFA mitigation zone" (the area within the 180-dB isopleth of the SURTASS LFA sonar source sound field) before and during low frequency transmissions. NMFS has also added an additional 1-kilometer (km) buffer zone beyond the LFA mitigation zone. This is shown in Figure 2.

Monitoring will (a) commence at least 30 minutes before the first SURTASS LFA sonar transmission; (b) continue between pings; and (c) continue for at least 15 minutes after completion of a SURTASS LFA sonar transmission exercise or, if marine mammals are showing abnormal behavior patterns, for a period of time until those behavior patterns return to normal or until conditions prevent continued observations.

The Navy proposes to use three monitoring techniques:

- Visual monitoring for marine mammals and sea turtles from the SURTASS LFA sonar vessel during daylight hours;

- Use of the passive (low frequency) SURTASS array to listen for sounds generated by marine mammals as an indicator of their presence; and

- Use of high frequency active sonar (High Frequency Marine Mammal Monitoring [HF/M3] sonar) to detect, locate, and track marine mammals (and possibly sea turtles) that might be affected by low frequency transmissions near the SURTASS LFA sonar vessel and the sound field produced by the SURTASS LFA sonar source array.

Visual Monitoring. Visual monitoring will include daytime observations from the SURTASS LFA sonar vessel for potentially affected species. This monitoring will begin 30 minutes before sunrise, for ongoing transmissions, or 30 minutes before SURTASS LFA sonar is deployed and continue until 30 minutes after sunset or until SURTASS LFA sonar array is recovered. Personnel trained in detecting and identifying marine animals will make observations from the vessel. At least one observer, qualified by NMFS, will train, test and evaluate other visual observers. If a marine mammal is detected within the 180-dB LFA mitigation zone or the 1 km (0.54 nm buffer zone extending beyond the LFA mitigation zone, SURTASS LFA sonar transmissions will be immediately suspended. Transmissions will not resume less than 15 minutes after:

1. All marine mammals have left the area of the LFA mitigation and buffer zones; and
2. There is no further detection of any marine mammal within the LFA mitigation and buffer zones as determined by the visual and/or passive or active acoustic monitoring.

Passive Acoustic Monitoring. Passive acoustic monitoring for low frequency sounds generated by marine mammals will be conducted when SURTASS is deployed. The following actions will be taken:

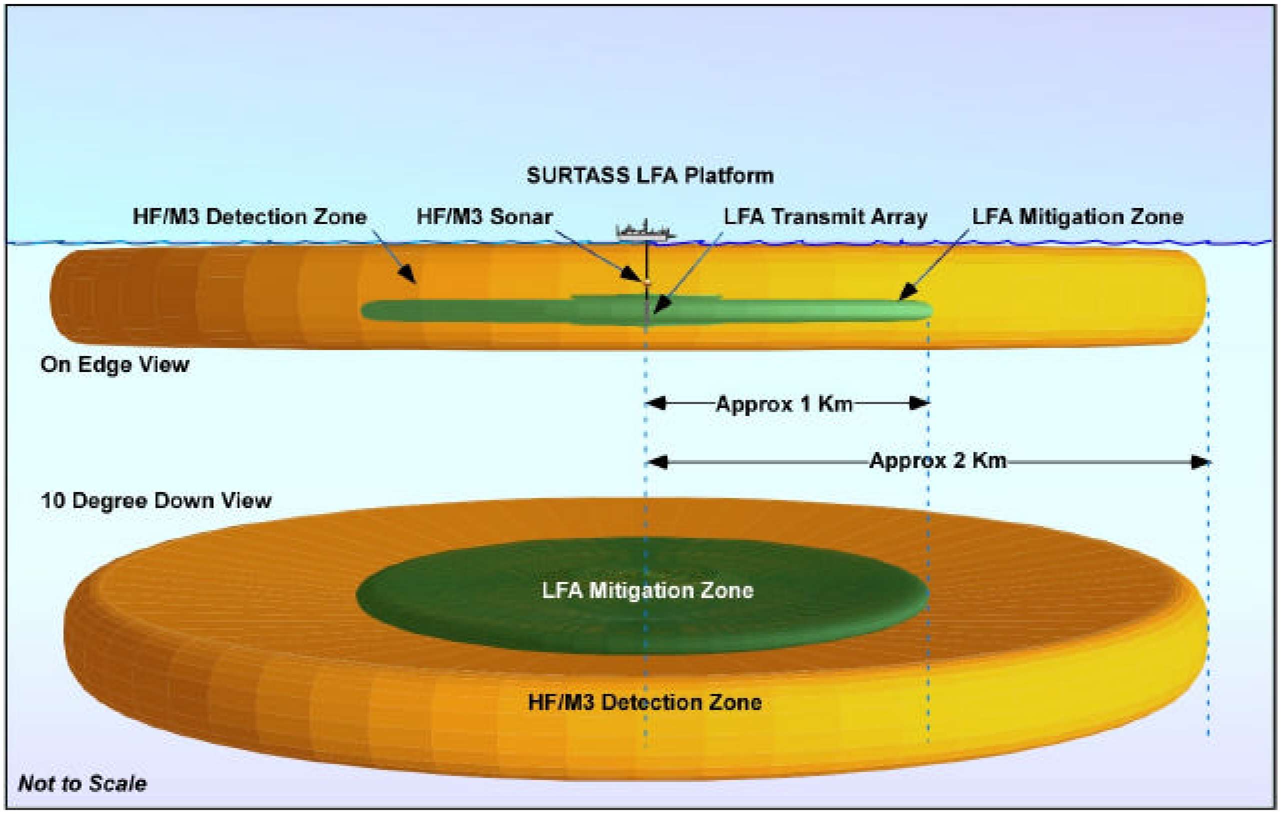

Figure 2. HF/M3 Sonar Detection and LFA Mitigation Zones

1. If sounds are detected and estimated to be from a marine mammal, the technician will notify the Officer in Charge who will alert the HF/M3 sonar operator and visual observers;
2. If a sound produced by a marine mammal is detected, the technician will attempt to locate the sound source using localization software; and
3. If it is determined that the animal will pass within the LFA mitigation zone or 1-km buffer zone (prior to or during transmissions), then the Officer in Charge will order the delay/suspension of transmissions when the animal is predicted to enter either of these zones.

The SURTASS passive horizontal line array is about 1,500 meters long (4,920 feet), has an operational depth from 152 m (500 ft) to 457 m (1,500 ft); and detects frequencies from 0 to 500 Hz.

High Frequency Active Acoustic Monitoring. The Navy will conduct high frequency active acoustic monitoring (by using an enhanced, commercial-type high frequency sonar) to detect, locate, and track marine mammals (and possibly sea turtles) that could pass close enough to the SURTASS LFA sonar transmit array to exceed the 180-dB mitigation criterion. This Navy-developed High Frequency Marine Mammal Monitoring (called HF/M3) sonar operates with a similar power level, signal type, and frequency as high frequency "fish finder" type sonars used worldwide by both commercial and recreational fishermen.

The HF/M3 source will be ramped-up slowly to operating levels over a period of no less than 5 minutes:

1. No later than 30 minutes before the first SURTASS LFA sonar transmission;
2. Prior to any SURTASS LFA sonar calibrations or tests that are not part of regular SURTASS LFA sonar transmissions; and
3. Anytime after the HF/M3 source has been powered down for a period of time greater than 5 minutes.

The HF/M3 source will not increase its sound pressure level once a marine mammal is detected; ramp-up may proceed once marine mammals are no longer detected.

HF/M3 Sonar, LFA Mitigation Zone, And Sound Propagation

The extent of the LFA mitigation zone (i.e., within the 180-dB sound field) is estimated by onboard acoustic modeling and environmental data collected *in situ*. Factored into this calculation are SURTASS LFA sonar source physical parameters of tow speed, depth, vertical steering, signal waveform/wavetrain selection, and peak transmit source level.

The HF/M3 sonar is located near the top of the SURTASS LFA sonar vertical line array. The HF/M3 sonar computer terminal for data acquisition/processing/display will be located in the SURTASS Operations Center. The HF/M3 sonar uses frequencies from 30 to 40 kHz with a variable bandwidth (1.5 to 6 kHz nominal); a 3-4 percent (nominal) duty cycle; a source level of

220 dB re 1 μPa (1 micropascal) at 1 m; a five-minute ramp-up period; and a maximum, nominal detection range of 2-2.5 km (1.08-1.35 nm).

The HF/M3 sonar will operate continuously while the SURTASS LFA sonar is deployed. A remote display from the PC control station will be situated at the Watch Supervisor console, which will be manned 24 hours a day during all SURTASS or SURTASS LFA sonar operations at sea.

When a marine animal is detected by the HF/M3 sonar, it automatically triggers an alert to the Watch Supervisor, who will notify the Officer in Charge. The Officer in Charge will then order the immediate delay/suspension of SURTASS LFA sonar transmissions until the animal is determined to have moved beyond this zone. All contacts will be recorded and provided as part of the long-term monitoring program associated with the proposed action.

Analysis and testing of the HF/M3 sonar operating capabilities indicate that this system substantially increases the probability of detecting marine mammals within the LFA mitigation zone. It also provides an excellent monitoring capability (particularly for medium to large marine mammals) beyond the LFA mitigation zone, out to 2 to 2.5 km (1.08 to 1.35 nm). Recent testing of the HF/M3 sonar, as documented in the SURTASS LFA Sonar Final EIS Subchapter 2.3.2.2, has demonstrated a probability of single-ping detection above 95 percent within the LFA mitigation zone for most marine mammals.

When the SURTASS LFA sonar is deployed, all marine mammal and sea turtle sightings/ detections would be recorded and provided as part of the Long Term Monitoring Program associated with the proposed action.

Geographic Restrictions

Both the Navy's proposed action and NMFS' proposed regulations restrict the Navy from operating SURTASS LFA sonar in a way that causes sonar sound fields to exceed 180 dB (re 1 μPa_{rms}) within 12 nautical miles (22 kilometers) of any coastline, including offshore islands, or designated offshore areas that are biologically important for marine mammals outside the 12 nautical mile (22 kilometer) zone during seasons specified for a particular area (see Table 1). When in the vicinity of known recreational and commercial dive sites, SURTASS LFA sonar will be operated to ensure that the sound field at these sites would not exceed 145 dB.

In addition, the Navy proposes to implement the following measures that were recommended by the NOAA Office of National Marine Sanctuaries to protective sanctuary resources: (1) in the Monterey NMS, received levels should not exceed 180 dB throughout the sanctuary; (2) in the Gulf of the Farallones and Cordell Bank National Marine Sanctuaries, received levels should not exceed 180 dB for areas that extend beyond 12 nm (22 km); (3) in the Olympic Coast National Marine Sanctuary, received levels in the sanctuary should not exceed 180 dB from the area from the shore to 23 nm (37.4 km) in the months of December, January, March and May; and (4) in the Hawaiian Islands Humpback Whale National Marine Sanctuary, received levels should not exceed 180 dB from November 1 through May 1.

NMFS' Proposed Small Take Authorization

As discussed previously, the National Marine Fisheries Service's Marine Mammal Conservation Division proposes to amend its regulations governing the take and importation of marine mammals (50 CFR Part 216) to make it possible for NMFS to issue annual letters of authorization that would allow the U.S. Navy to take marine mammals incidental to the U.S. Navy's operations of SURTASS LFA sonar for a five-year period ending in 2007. The proposed regulations would allow the U.S. Navy to incidentally, but not intentionally, take marine mammals by harassment and injury within the action area (see the delineation of the Action Area, which follows). NMFS proposes to take this action pursuant to section 101(a)(5)(A) of the MMPA (16 U.S.C. 1361 *et seq*.). The Secretary may grant permission for periods up to 5 years upon finding that the taking will be small, have a negligible impact on the species or stock(s) of affected marine mammals, and will not have an unmitigatible adverse impact on the availability of the species or stock(s) for subsistence uses, and if regulations are prescribed setting forth the permissible methods of taking and the requirements pertaining to the monitoring and reporting of such taking.

Table 1. Offshore areas of biological importance to marine mammals

Area	Location	Important Months	Important for
200-meter isobath off the North American Coast	From 28° N to 50° N, west of 40° W	Year round	right whale (western Atlantic stock), sei whale, humpback whale, northern bottlenose whale
Antarctic Convergence Zone	30° E to 80° N : 45° S 80° E to 150° E: 55° S 150° E to 50° W: 60° S 50° W to 30° E: 50° S	October through March	blue whale, fin whale, sei whale, minke whale, humpback whale, sperm whale, killer whale, southern bottlenose whale, Arnoux's beaked whale, Gray's beaked whale, strap-toothed beaked whale, Commerson's dolphin, Peale's dolphin, hourglass dolphin, dusky dolphin
Costa Rica Dome	Centered at 9° N and 88° W	Year round	blue whale, olive ridley sea turtle
Penguin Bank	South of Molokai Island, Hawaii	November 1 through May 1	Humpback whale

Action Area

Figure 3 shows the geographic areas in which SURTASS LFA could operate. The action area for this biological opinion encompasses the Atlantic, Pacific, and Indian Oceans, Mediterranean Sea, and associated seas, including the Arabian Sea, Barents Sea, Caribbean Sea, Norwegian Sea, Sea of Okhotsk, Phillipine Sea, and Tasman Sea. Specific areas include:

(1) North Atlantic Ocean, including (i) Western North Atlantic, from 35° N lat. north to a line between Cape Chidley, Labrador northeast to Nuuk, Greenland, and from the North American continent east to 41° W longitude, (ii) Eastern North Atlantic, from 35° N lat.

north to 72° N lat. and 41° W long. east to the European continent;

(2) Mediterranean Sea;

(3) North Pacific Ocean, including (i) Western North Pacific, from 20° N lat. north to the Aleutian Island chain and the Sea of Okhotsk, and from the Asian continent east to 175° W longitude, (ii) Eastern North Pacific, from 42° N lat. north to Alaska and the south side of the Aleutian Islands and from the North American continent west to 175° W longitude;

(4) Central Atlantic Ocean, including (i) Eastern Central Atlantic, from 7° S lat. north to 35° N lat. and from the African continent west to 40° W long. between 5° N lat. and 35° N lat., to 30° W long. between 0° lat. and 5° N lat., to 20° W long. between 7° S lat. and 0° latitude, and from 20° W longitude to the African coast along the 7° S lat. (ii) Western Central Atlantic, from 5° N lat. north to 35° N lat., and from the American continent, east to 40° W longitude;

(5) Indian Ocean, including (i) Eastern Indian Ocean, from 60° S lat. north to the Bay of Bengal, and Asian continent, and from 80° E long. east to the Asian continent, the Sunda Islands and Australia and to 150° E longitude, (ii) Western Indian Ocean, from 60° S lat. north to the Arabian Sea, and from 30° E long. east to 80° E longitude;

(6) Central Pacific Ocean, including (i) Western Central Pacific, from 175° W long., east to the Asian continent and Indonesia, and from 10° S lat., north to 20° N lat., (ii) Central Pacific, from 10° S lat., north to 42° N lat. between 175° W long. and 130° longitude, (iii) Eastern Central Pacific, from 5° S lat. north along the American coastline to 42° N lat., from 130° W long. along 10° S lat. to 105° W long., from 10° S lat. along 105° W long. to 5° S lat., from 105° W long. along 5° S lat. to the South American coastline, from 130° W long. along 42° N lat. to the North American coastline and from 42° N lat. to 10° S lat. along the 130° W longitude;

(7) South Pacific Ocean, including (i) Western South Pacific from 60° S lat. north to 10° S lat. and from the east coast of Australia in the north and 150°E long. south of Australia east to 105° W longitude, (ii) Eastern South Pacific from 60° S lat. north to 5° S lat. and from the 105° W long. east to the South American coastline in the north and 70° W long. in the south;

(8) South Atlantic Ocean, (i) Western South Atlantic, from 60° S lat. north to 5° N lat. in the area west of 30° W long., and from 60° S lat. north to 0° lat. in the area east of 30° W long. and from the South American continent east to 30° W long. between 0° and 5° N lat. and east to 20° W long. between 0° and 60° longitudes latitude and (ii) East South Atlantic from 50° S lat. north to 7° S lat. and from 20° W long. east to the African coastline in the north and 30° E long. south of the continent.

The proposed action may occur in any marine waters within these areas that are (1) more than 12 nautical miles (22 kilometers) from shore and (2) are not within one of the offshore biologically important areas listed in Table 1, subject to the 145-dB and 180-dB restrictions delineated above.

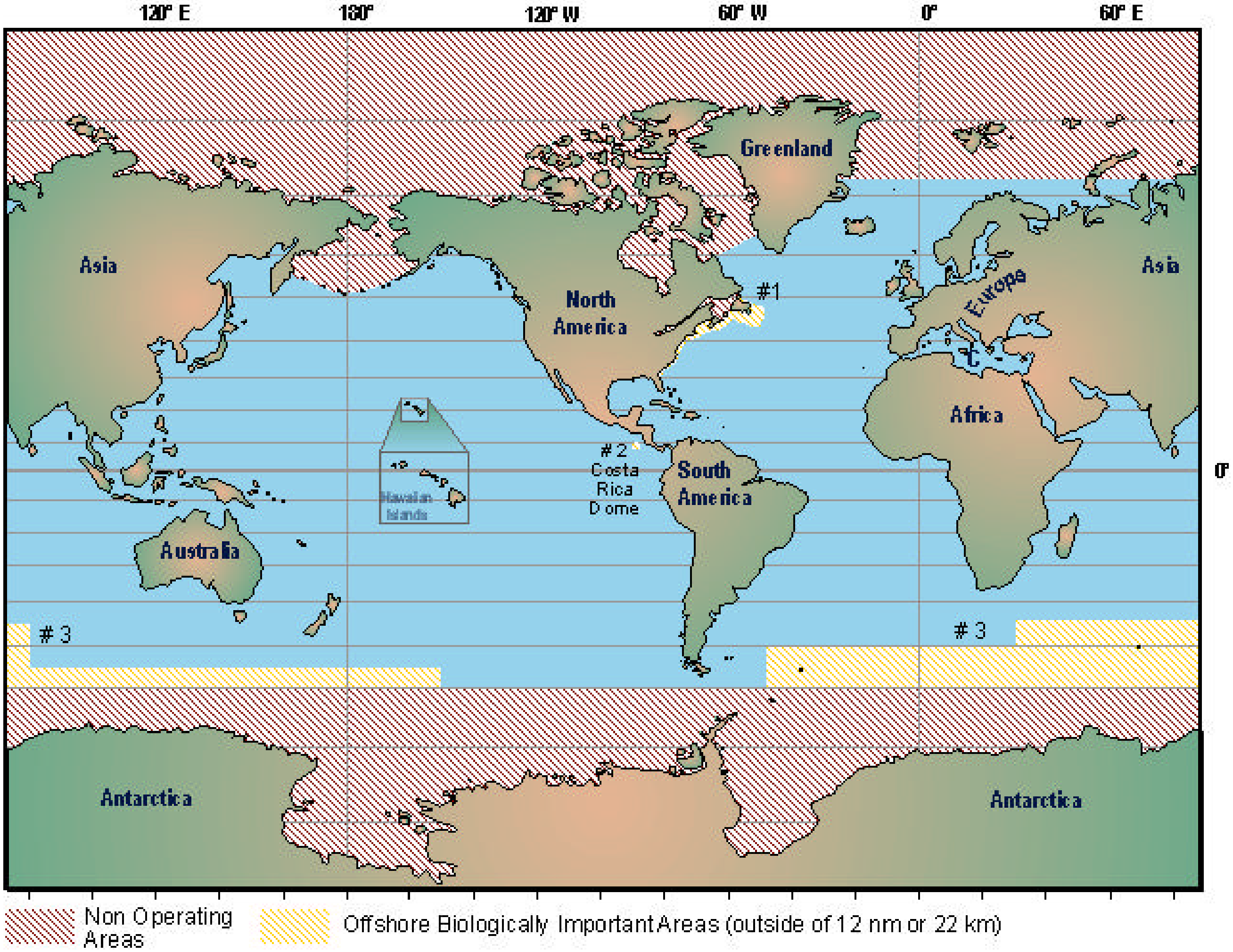

Figure 3. Geographic areas in which SURTASS LFA could operate (because of the scale, all offshore biologically important areas may not appear on this map)

STATUS OF THE SPECIES AND CRITICAL HABITAT

NMFS has determined that the actions being considered in this biological opinion may affect the following species and critical habitat, under NMFS' jurisdiction, that are protected under the Endangered Species Act of 1973 (16 U.S.C. 1531 *et seq*.; ESA)[2]:

Steller sea lion (western population)	*Eumetopias jubatus*	Endangered
Steller sea lion (eastern population)		Threatened
Caribbean monk seal	*Monachus tropicalis*	Endangered
Guadalupe fur seal	*Arctocephalus townsendi*	Threatened
Hawaiian monk seal	*Monachus schausinslandi*	Endangered
Mediterranean monk seal	*Monachus monachus*	Endangered
Blue whale	*Balaenoptera musculus*	Endangered
Bowhead whale	*Balaena mysticetus*	Endangered
Fin whale	*Balaenoptera physalus*	Endangered
Gray whale (Western Pacific population)	*Eschrichtius robustus*	Endangered
Humpback whale	*Megaptera novaeangliae*	Endangered
Right whale	*Eubalaena glacialis (including australis)*	Endangered
Sei whale	*Balaenoptera borealis*	Endangered
Sperm whale	*Physeter macrocephalus*	Endangered
Green sea turtle	*Chelonia mydas*	Threatened Endangered
Hawksbill sea turtle	*Eretmochelys imbricata*	Endangered
Kemp's ridley sea turtle	*Lepidochelys kempii*	Endangered
Leatherback sea turtle	*Dermochelys coriacea*	Endangered
Loggerhead sea turtle	*Caretta caretta*	Threatened
Oliver ridley sea turtle	*Lepidochelys olivacea*	Threatened Endangered
Atlantic salmon (Gulf of Maine)	*Salmo salar*	Endangered
Chinook salmon (Puget Sound)	*Oncorhynchus tshawytscha*	Threatened
Chinook salmon (Lower Columbia River)		Threatened
Chinook salmon (Upper Columbia River Spring)		Endangered
Chinook salmon (Upper Willamette River)		Threatened
Chinook salmon (Central Valley spring)		Threatened
Chinook salmon (Sacramento River winter)		Endangered
Chinook salmon (Snake River spring/summer)		Threatened
Chinook salmon (Snake River fall)		Threatened
Chum salmon (Columbia River)	*Oncorhynchus keta*	Threatened
Chum salmon (Hood Canal summer run)		Threatened
Coho salmon (Central California Coast)	*Oncorhynchus kisutch*	Threatened
Coho salmon (Oregon Coast)		Threatened

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[2] Species are listed according to the order in which they appear in the list of endangered and threatened fish and wildlife and plants (50 CFR Part 17).

Coho salmon (Southern Oregon Northern Coastal California)		Threatened
Sockeye salmon (Ozette Lake)	*Oncorhynchus nerka*	Endangered
Sockeye salmon (Snake River)		Endangered
Steelhead (Upper Columbia River)	*Onchorynchus mykiss*	Endangered
Steelhead (Middle Columbia River)		Threatened
Steelhead (Lower Columbia River)		Threatened
Steelhead (Upper Willamette River)		Threatened
Steelhead (Snake River Basin)		Threatened
Steelhead (Northern California)		Threatened
Steelhead (California Central Valley)		Threatened
Steelhead (Central California Coastal)		Threatened
Steelhead (South Central California)		Threatened
Steelhead (Southern California)		Threatened
Gulf sturgeon	*Acipenser oxyrinchus desotoi*	Threatened
Shortnose sturgeon	*Acipenser brevirostrum*	Endangered

Designated critical habitat

Steller sea lion	portions of the north Pacific Ocean
Monk seal	portions of the north Pacific Ocean
Right whale	portions of the western Atlantic Ocean
Green sea turtle	portions of the Caribbean Sea
Hawksbill sea turtle	portions of the Caribbean Sea
Leatherback sea turtle	portions of the Caribbean Sea

Proposed Species

Smalltooth sawfish	*Pristic pectinata*	Proposed Endangered

NMFS also recognizes that gray whales (*Eschrictius robustus*) migrate through the action area during their spring and fall migrations toward the Chukchi and Beaufort Seas. Although the eastern Pacific population of the gray whale was removed from the list of threatened and endangered species in 1994 (59 FR 31094), the Western Pacific population remains endangered. NMFS has a continuing obligation to monitor the status of this species. This biological opinion will not assess whether the effects of SURTASS LFA sonar are likely to jeopardize the continued existence of the eastern Pacific population of gray whales; however, this opinion will include a general assessment of the effects of the action on this population of gray whales as part of NMFS' continuing responsibility to monitor the status of the species.

Species and Critical Habitat Not Discussed Further in This Biological Opinion

To assess the effects of SURTASS LFA on threatened and endangered species and critical habitat, NMFS used a generalized assessment model patterned after the models toxicologists and epidemiologists use to assess risks posed by terrestrial, aquatic, and atmospheric pollutants (Lipton *et al*. 1993, U.S. EPA 1996; also see further discussion in the *Effects of the Action* section of this Opinion).

The first step in our approach assessed the likelihood of a species or critical habitat being exposed to sound pressure levels associated with SURTASS LFA sonar, including an assessment of the intensity, duration, and frequency of any exposure. For species or critical habitat that were likely to be exposed to SURTASS LFA sonar, the second step of our approach assessed probable ecological responses of listed species to SURTASS LFA sonar or, alternatively, the potential effects of differing levels of low-frequency sound on listed species based on their susceptibility to sound pressure levels and frequencies associated with the SURTASS LFA sound source and their potential responses to those levels (that is, "stressor-response" relationships). For species or critical habitat that were likely to be exposed to SURTASS LFA sonar and, if exposed, were susceptible to SURTASS LFA sonar, the third step of our approach estimated the probable risks posed by SURTASS LFA sonar based on the exposure estimates and susceptibilities to reach conclusions about the potential effects of SURTASS LFA sonar on species and critical habitat (also known as "risk characterization").

The preceding list of threatened and endangered species and designated critical habitat includes listed species and critical habitat that generally occur in the oceans that may be affected by SURTASS LFA. Although formal consultation is required for actions that "may affect" listed species or designated critical habitat, the Services normally consult formally only on actions that have already undergone informal consultation and, therefore, are likely to adversely affect listed species or designated critical habitat (see 50 CFR 402.14(a) for further discussion of these differences). To refine the scope of this biological opinion, NMFS used two criteria (risk factors) to screen species or critical habitat that are not likely to be adversely affected by SURTASS LFA sonar. The first criterion was *exposure*: species that are not likely to be exposed to SURTASS LFA sonar (at sound pressure levels at or below ambient) are not likely to be adversely affected by the sonar. The second criterion was *susceptibility* upon exposure: species that may be exposed to sound transmissions from the sonar, but are likely to be unaffected by the sonar (at sound pressure levels they are likely to be exposed to) are also not likely to be adversely affected by the sonar. This section summarizes the results of those evaluations.

Threatened and Endangered Pinnipeds

Mediterranean monk seals are found in several fragmented and isolated stocks throughout their former range in the Mediterranean and Black Seas, and the Atlantic coast and offshore islands of North Africa. Mediterranean monk seals tend to stay close to their haul-out areas and forage in coastal waters for fish, octopus, and crustaceans. Mediterranean monk seals forage in water less than 70 m (230 ft) deep. Because of their ecology and biogeography, the operational rules that restrict the Navy from operating SURTASS LFA sonar in a way that causes sonar sound fields to exceed 180 dB within 12 nautical miles (22 kilometers) of any coastline, including offshore islands; and high bottom and surface signal losses that would prevent SURTASS LFA signals from penetrating coastal waters with appreciable signal strengths; Mediterranean monk seals are not likely to be exposed to sound pressure levels from SURTASS LFA sonar and, therefore, are not likely to be adversely affected by the sonar.

There have been no confirmed sightings of Caribbean monk seals (also known as the West Indian monk seal) since 1952. This species is now considered extinct despite periodic suggestions that these shy animals may persist in isolated locations of the Caribbean. Nevertheless, they will not be considered further in this Opinion until their existence has been verified.

Because of these determinations, Mediterranean and Caribbean monk seals will not be considered further in this biological opinion.

Threatened and Endangered Fish Species

Shortnose sturgeon are an anadromous species that occurs along the Atlantic Coast of North America, from the St. John River in Canada to the St. John's River in Florida. The recovery plan for shortnose sturgeon recognized 19 distinct, wild populations: New Brunswick, Canada (1 population); Maine (2 populations); Massachusetts (1 population); Connecticut (1 population); New York (1 population); New Jersey and Delaware (1 population); Maryland and Virginia (1 population); North Carolina (1 population); South Carolina (4 populations); Georgia (4 populations); and Florida (2 populations). One partially-landlocked population occurs in Holyoke Pool of the Connecticut River. Another landlocked population may exist in Lake Marion on the Santee River in South Carolina. Because of their ecology, biogeography, the operational rules that restrict the Navy from operating SURTASS LFA sonar in a way that causes sonar sound fields to exceed 180 dB in offshore biologically important areas (which includes the 200-meter isobath off the North American coast), and high bottom and surface signal losses that would prevent LFA signals from penetrating coastal waters with appreciable signal strengths, shortnose sturgeon are not likely to be exposed to sound pressure levels from SURTASS LFA sonar and, therefore, are not likely to be adversely affected by the sonar.

The Gulf sturgeon is also an anadromous species that spends the majority of its life in large, fresh water, coastal rivers on the Gulf of Mexico. Because of their ecology, biogeography, and the operational rules that restrict the Navy from operating SURTASS LFA sonar in a way that causes sonar sound fields to exceed 180 dB in offshore biologically important areas (which includes the 200-meter isobath off the North American coast), gulf sturgeon are not likely to be exposed to sound pressure levels from SURTASS LFA sonar and, therefore, are not likely to be adversely affected by the sonar.

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). On April 16, 2001, NMFS proposed to list smalltooth sawfish as endangered under the ESA.

Like the sturgeon, the ecology and biogeography of smalltooth sawfish, the operational rules that restrict the Navy from operating SURTASS LFA sonar in a way that causes sonar sound fields to exceed 180 dB in offshore biologically important areas (which includes the 200-meter isobath off the North American coast), and high bottom and surface signal losses that would prevent LFA signals from penetrating coastal waters with appreciable signal strengths, smalltooth sawfish are not likely to be exposed to sound pressure levels from SURTASS LFA sonar and, therefore, are not likely to be adversely affected by the sonar.

Because of these determinations, shortnose sturgeon, Gulf sturgeon, and smalltooth sawfish will not be considered further in this biological opinion.

Critical Habitat

Critical habitat for the eastern and western populations of Steller sea lions includes all major Steller sea lion rookeries and major haulouts and terrestrial, atmospheric, and aquatic zones associated with those rookeries and haulouts. For the eastern population of Steller sea lions, the terrestrial zone extends 3,000 ft (0.9 km) landward from the baseline or base point of each major rookery and major haulout; the air zone extends 3000 ft (0.9 km) above the terrestrial zone of each major rookery and major haulout measured vertically from sea level; the aquatic zone extends 3,000 ft (0.9 km) seaward in State and Federally managed waters from the baseline or basepoint of each major haulout in Alaska that is east of 144° W longitude. Critical habitat includes an aquatic zone that extends 20 nm (37 km) seaward in State and Federally managed waters from the baseline or basepoint of each major rookery and major haulout in Alaska that is west of 144° W long.

Critical habitat has also been designated for the northern right whale in the Atlantic Ocean in Cape Cod Bay, Great South Channel, and off Georgia and Florida (50 CFR 226.13).

In May 1988, NMFS designated critical habitat for the Hawaiian monk seal from shore out to 20 fathoms in 10 areas of the northwestern Hawaiian Islands. Critical habitat for these species includes “all beach areas, sand spits and islets, including all beach crest vegetation to its deepest extent inland, lagoon waters, inner reef waters, and ocean waters out to a depth of 20 fathoms around the following: Kure Atoll, Midway Islands, except Sand Island and its harbor, Lisianski Island, Laysan Island, Maro Reef, Gardner Pinnacles, French Frigate Shoals, Necker Island, and Nihoa Island” (50 CFR 226.201).

Based on the best scientific and commercial data available, SURTASS LFA sonar transmissions may affect, but are not likely to adversely affect, critical habitat for right whales, Steller sea lions, and Hawaiian monk seals because the SURTASS LFA sonar will remain far enough from shore to limit sound levels to below 180 dB within 12 nm (22 km) of land and the 200-m (656-ft) isobath of the North American east coast. Critical habitat for Steller sea lions and right whales consists of coastal waters where received levels will be below 180 dB. Further, the Navy has included the critical habitat for right whales in its areas of biological importance, which provides that critical habitat with additional buffers. For these reasons, NMFS concludes that Navy's employment of SURTASS LFA sonar is not likely to adversely affect this critical habitat. Therefore, critical habitat will not be considered further in this biological opinion.

Introduction to the Status of Species Considered in this Opinion

The remaining discussions in this section of the Opinion will focus on the status of the threatened and endangered species that occur in the action area and that may be adversely affected by SURTASS LFA sonar. The information presented in this section is a summary of the information available and is necessary to establish the environmental baseline for each species, which will be used as the foundation for the effects analyses presented in the *Effects of the Action* section of this Opinion. More detailed background information on the status of these species and critical habitat can be found in a number of published documents including recent recovery plans for sea turtles (NMFS and USFWS 1998a, 1998b, 1998c, 1998d, and 1998e) status reviews, recovery plans for the humpback whale (NMFS 1991a), right whale (NMFS 1991b), blue whale (NMFS 1998c), fin and sei whale (NMFS 1998), Steller sea lion (NMFS 1995), and a whale status report prepared by Perry *et al*. (1999). Richardson *et al.* (1995) and Tyack (2000) provide detailed analyses of the functional aspects of cetacean communication. Finally, Croll *et al*. (1999) and NRC (1994, 1996, 2000) provide information on the potential and probable effects of low frequency sound on the marine vertebrates considered in this Opinion.

Because the proposed action could occur in the Atlantic Ocean, Indian Ocean, Pacific Ocean, or Mediterranean Sea, the action area encompasses the entire range of virtually all of the species discussed in the following section. Consequently, the *Status of the Species* section of this Opinion is the same as the *Environmental Baseline*, so the two sections have been combined into a single discussion.

Steller Sea Lion

Species description and distribution

Steller sea lions are distributed around the rim of the North Pacific Ocean from the Channel Islands off Southern California to northern Hokkaido, Japan. In the Bering Sea, the northernmost major rookery is on Walrus Island in the Pribilof Island group. The northernmost major haulout is on Hall Island off the northwestern tip of St. Matthew Island. Their distribution also extends northward from the western end of the Aleutian chain to sites along the eastern shore of the Kamchatka Peninsula. Their distribution is probably centered in the Gulf of Alaska and the Aleutian Islands (NMFS 1992).

Within their range, land sites used by Steller sea lions are referred to as rookeries and haulouts. Rookeries are used by adult sea lions for pupping, nursing, and mating during the reproductive season (generally from late May to early July). Haulouts are used by all ages classes of both genders but are generally not where sea lions reproduce. The continued use of particular sites may be due to site fidelity, or the tendency of sea lions to return repeatedly to the same site, often the site of their birth. Presumably, these sites were chosen and continue to be used because they protect sea lions from predators, some measure of protection from severe climate or sea surface conditions, and (perhaps most importantly) are in close proximity to prey resources.

The movement patterns of Steller sea lions are not yet well understood but we can provide a general picture of the information we have. Sea lions move on and offshore for feeding excursions. At the end of the reproductive season, some females may move with their pups to other haulout sites and males may “migrate” to distant foraging locations (Spaulding 1964). Sea lions may make semi-permanent or permanent one-way movements from one site to another (Chumbley *et al*. 1997, their Table 8; Burkanov *et al*. unpubl. report [cited in Loughlin 1997]). Calkins and Pitcher (1982) reported movements in Alaska of up to1500 km. They also describe wide dispersion of young animals after weaning, with the majority of those animals returning to the site of birth as they reach reproductive age.

Life history information

Steller sea lions have a polygynous reproductive strategy in which a single male may mate with multiple females. As mating occurs on land (or in the surf or intertidal zones), males are able to defend territories and thereby exert at least partial control over access to adult females and mating privileges. The pupping and mating season is relatively short and synchronous, probably due to the strong seasonality of the sea lions' environment and the need to balance aggregation for reproductive purposes with dispersion to exploit distant food resources (Bartholomew 1970). In May, adult males compete for rookery territories. In late May and early July, adult females arrive at the rookeries, where pregnant females give birth to a single pup. Mating occurs about one to two weeks later (Gentry 1970). The gestation period is probably about 50 to 51 weeks, but implantation of the blastocyst is delayed until late September or early October (Pitcher and Calkins 1981). Due to delayed implantation, the metabolic demands of a developing fetus are not imposed until well after fertilization. The sex ratio of pups at birth is assumed to be about 1:1 (e.g., York 1994) or biased toward slightly greater production of males (e.g., Pike and Maxwell 1958, NMFS 1992).

For females with a pup, the nursing period continues for months to several years. The transition to nutritional independence may, therefore, occur over a period of months as the pup begins to develop essential foraging skills, and depends less and less on the adult female. The length of the nursing period may also vary as a function of the condition of the adult female. The nature and timing of weaning is important because it determines the resources available to the pup during the more demanding winter season and, conversely, the demands placed on the mother during the same period. The maintenance of the mother-offspring bond may also limit their distribution or the area used for foraging.

The reproductive cycle includes mating, gestation, parturition, and nursing or post-natal care. The adult female's ability to complete this cycle successfully depends largely on the prey available to her. While much of the effort to explain the Steller sea lion decline has focused on juvenile survival rates, considerable evidence suggests that the decline may also be due, in part, to decreased reproductive success by adult females.

Diving and social behavior

Kenyon (1952) reported that Steller sea lions were hooked on fishing lines at depths of 183 meters. Unpublished information from NMFS' National Marine Mammal Laboratory suggests that Steller sea lions generally feed at shallow depths, but will dive to depths of 277 meters.

Because of their polygynous breeding behavior, in which individual, adult male sea lions will breed with a large number of adult females, Steller sea lions have clearly-defined social interactions. As a result, Steller sea lions are gregarious on rookeries and haulouts and are often found in groups at sea. King (1983 in Croll *et al.* 1999) reported rafts of several hundred Steller sea lions adjacent to haulouts.

Vocalizations and hearing

Gentry (1970) and Sandegren (1970) described a suite of sounds that Steller sea lions form while on their rookeries and haulouts. These sounds include threat displays, vocal exchanges between mothers and pups, and a series of roars and hisses. Poulter (1971) reported that Steller sea lions produce clicks, growls, and bleats underwater.

Listing status

Steller sea lions were listed as threatened under the Endangered Species Act on November 26, 1990 (55 FR 49204). The listing followed a decline in the U.S. population of about 64% over the three decades prior to the listing. In 1997, the species was split into two separate populations based on demographic and genetic differences (Bickham *et al*. 1996, Loughlin 1997), the western population was reclassified to endangered while the eastern population remained threatened (62 FR 30772).

Population status and trends

Numbers of Steller sea lions declined dramatically throughout much of the species' range, beginning in the mid- to late 1970s (Braham *et al*. 1980, Merrick *et al*. 1987, NMFS 1992, NMFS 1995). For two decades prior to the decline, the estimated total population was 250,000 to 300,000 animals (Kenyon and Rice 1961, Loughlin *et al*. 1984). The population estimate declined by 50-60% to about 116,000 animals by 1989 (NMFS 1992), and by an additional 15% by 1994 (Sease *per. com.*).

The decline has been restricted to the western population of Steller sea lions which has declined by about 5% per year during the 1990s. Counts for this population have fallen from 109,880 animals in the late 1970s to 22,167 animals in 1996, a decline of 80% (NMFS 1995). Although the number of animals lost appears to have been far greater from the late 1970s to the early 1990s, the rate of decline has remained high. The 1996 count was 27% lower than the count in 1990. Final results from counts conducted in 1998 are not yet available, but preliminary results for trend sites between the Kenai Peninsula to Kiska Island indicate a decline of about 9% in nonpups since 1996, and 19% in pups since 1994.

During this same time, the eastern stock has remained stable or increased by several percent per year, in Southeast Alaska (Sease and Loughlin 1999), in British Columbia, Canada (P. Olesiuk, Department of Fisheries and Oceans, unpubl. data), and in Oregon (R. Brown, Oregon Department of Fish and Wildlife, unpubl. data). Approximately 60% of Steller sea lions belong to the western stock, 40% to the eastern stock (Sease *et al*., in press). Counts in Russian territories have also declined and are currently estimated to be about one-third of historic levels (NMFS 1992).

Population viability analyses have been conducted by Merrick and York (1994) and York *et al*. (1996). The results of these analyses indicate that the next 20 years may be crucial for the Steller sea lion, if the rates of decline observed in 1985 to 1989 or 1994 continue. Within this time frame, it is possible that the number of adult females in the Kenai-to-Kiska region could drop to less than 5000 individuals. Extinction rates for rookeries or clusters of rookeries could increase sharply in 40 to 50 years, and extinction for the entire Kenai-to-Kiska region could occur in the next 100-120 years.

Impacts of human activity on this species

Of the two listed populations of Steller sea lions, the western population has the greatest risk of extinction. The endangered western population of Steller sea lions has declined by about 90 percent since the early 1970s and continues to decline throughout its range. This population is declining for many reasons and may now face threats that are different from the ones that caused the population's initial decline. From the 1950s through the 1980s, animals from this population were killed intentionally and unintentionally by fishers, in commercial harvests, and in subsistence harvests which may have begun to destabilize the population. The harvest of over 45,000 pups from 1963 to 1972 probably changed the number of animals that recruited into the adult, breeding population in that region and contributed to local population trends in the 1960s through the early 1980s in the Gulf of Alaska and the eastern Aleutian Islands. Similarly, subsistence harvests prior to the 1990s were not measured but may have contributed to population decline in localized areas where such harvests were concentrated.

Harassment has occurred in many areas and may have been very disruptive to sea lion colonies on rookeries or haulouts, thereby leading to redistribution or deaths of animals. Such harassment could have contributed to mortality if animals were shot, females were separated from their pups for long periods, or animals (especially pups) were trampled or crushed or otherwise injured in the stampedes that often accompany such harassment. Nevertheless, harassment is thought to be less common at present, and the data are not sufficient to demonstrate that harassment was a significant contributor to the decline. Harassment is also a less likely explanation in the remote areas of the sea lion range where declines have, nonetheless, been observed (e.g., central and western Aleutian Islands).

At the same time, portions of the North Pacific Ocean have undergone major changes in temperatures that have probably contributed to a shift in the trophic structure of the fish community in the Aleutian Islands, Bering Sea, and Gulf of Alaska. This shift may explain the shift from marine systems dominated by herring and capelin to systems dominated by pollack and flatfish. At the same time, the Bering Sea, Aleutian Islands, and Gulf of Alaska ecosystems have experienced the development and expansion of major fisheries for essential sea lion prey. The fisheries have also contributed to changes in the trophic structure of these ecosystems, but as is the case with natural changes, the extent of fisheries-related effects on the ecosystems at large can not be determined. With respect to Steller sea lions, however, fisheries target important prey resources at times and in areas where sea lions forage. The actual causes or the contribution of multiple causes has been, and continues to be, subject to extensive debate.

In the face of all these changes and influencing factors, the western population of Steller sea lions has not been able to maintain itself. The available evidence suggests that a significant part of the problem is lack of available prey. Studies of animals collected in the Gulf of Alaska in 1975-1978 and 1985-1986 indicate that animals in the latter collection were smaller, took longer to reach reproductive maturity, produced fewer offspring, tended to be older, and exhibited signs of anemia. In addition, survival of juvenile animals appeared to have dropped in both the eastern

Aleutian Islands (Ugamak Island; Merrick *et al*. 1987) and the Gulf of Alaska (Marmot Island; Chumbley *et al*. 1997).

Population viability analyses have been conducted by Merrick and York (1994) and York *et al*. (1996). The results of these analyses indicate that the next 20 years may be crucial for the western population of Steller sea lions, if the rates of decline observed in 1985 to 1989 or 1994 continue. Within two decades, it is possible that the number of adult females in the Kenai-to-Kiska region could drop to less than 5,000. Once the western population of Steller sea lions crosses this threshold, the small population size, by itself, could accelerate the population's decline to extinction. Extinction rates for rookeries or clusters of rookeries could increase sharply in 40 to 50 years and Steller sea lions could become extinct throughout the entire Kenai-to-Kiska region in the next 100-120 years. Based on these analyses, it is not reasonable to expect the western population of Steller sea lions to survive the various human-caused threats that led to their listing as an endangered species if these threats are not abated in the immediate future.

Guadalupe Fur Seal

Species description and distribution

Guadalupe fur seals are found on Guadalupe Island (Mexico) in the eastern Pacific Ocean off Mexico; a few individuals have been known to range as far north as Sonoma County, California, south to Los Islotes Islands in Baja California, Mexico. A few Guadalupe fur seals occupy California sea lion rookeries in the Channel Islands of California (Stewart *et al.* 1987 in Reeves *et al.* 1992).

Life history information

Very little is known about the population ecology and demography of Guadalupe fur seals. Females give birth to a single pup in June and pups are nursed until they are yearlings (Reeves *et al.* 1992). Adult males and adult female Guadalupe fur seals tend to return to the same rookeries each year, although only about 33 percent of the males held territories in sequential years (Gallo 1994 in Reeves *et al.* 1992). Although only limited information is available on their diets, they have been reported to forage on squid, lanternfish, octopus, and crustaceans (Reeves *et al.* 1992).

Listing status

Guadalupe fur seals were listed as threatened under the Endangered Species Preservation Act of 1966 on March 11, 1967. This listing was extended in 1973 under the Endangered Species Act of 1973.

Population status and trends

The size of this fur seal population prior to commercial exploitation is unknown, although several authors have suggested that their size ranged between 20,000 and 100,000 individuals. In 1993, their population was estimated to number about 7,300 animals (Gallo-Reynoso 1994).

Diving and social behavior

Guadalupe fur seals are shallow divers that forage in the upper 20 to 30 meters of the water column. They have mean dive depths of about 17 meters (for lactating females), with modal depths of 3.1 meters (Gallo-Reynoso 1994). The mean duration of their dives was 2.6 minutes. Like other otariids, Guadalupe fur seals have are social breeders: a single male will breed with several females.

Vocalizations and hearing

Peterson *et al.* (1968 in Croll *et al.* 1999) described their vocalizations as barks, roars, and coughs.

Impacts of human activity on this species

Guadalupe fur seals were driven close to extinction by sealers, sea otter hunters, and whalers between the late 1700s and early 1800s. By 1897, they were believed to be extinct. In the 1920s, small number of these seals were rediscovered on Guadalupe Island, which appeared to have been their sole, remaining rookery. In the 1950s, less than 20 fur seals were known to exist. Since the 1950s, these fur seals have increased in abundance by about 10 percent per year; by the mid-1970s, they had increased to about 1,000 animals; by the mid-1980s, they had increased to about 3,300 animals (Gallo-Reynoso 1994) and appear to have remained at this level since (Barlow *et al*. 1997). Information on contemporary threats to Guadalupe fur seals remains largely unknown, although they may be captured and killed in gillnet fisheries in Mexico and the United States (Barlow *et al.* 1997).

Hawaiian Monk Seal

Species description and distribution

The Hawaiian monk seal is found primarily on the Leeward Chain of the Hawaiian Islands, especially Nihoa, Necker, French Frigate Shoals, Pearl and Hermes Reef, Kure Atoll, Laysan, and Lisianski. Sightings on the main Hawaiian Islands have become more common in the past 15 years and a birth was recorded on Kauai and Oahu in 1988 and 1991 respectively (Kenyon 1981, Riedmann 1990). Midway was an important breeding rookery, but is no longer used (Reeves *et al.* 1992). Hawaiian monk seals breed primarily at Laysan Island, Lisianski Island, and Pearl and Hermes Reefs (Tomich 1986). Monk seals are increasingly sighted in the main Hawaiian Islands. Additional sightings have occurred on at least three occasions at the remote Pacific location of Johnston Island (excluding nine adult males translocated from Laysan Island in 1984) over the past 30 years.

Life history information

Monk seals tend to stay near land (Tomich 1986), and small numbers (1-4) are regularly seen around Kauai and each of the other main Hawaiian Islands (Nitta, pers. comm. 1995). There is also a small population on the island of Niihau. Most pups are born between March and May, but pupping has been recorded year-round (U.S. Dept. of Commerce 1986). A single female gave birth to a female pup on the north coast of Kauai in 1988 (Reeves *et al.* 1992) and a pup was born in the Poipu Beach area during the summer of 1989 (Naughton, pers. comm. 1990). In the summer of 2000, two pups were born at the south shore Poipu area, Kauai, and an additional pup was born on the north shore of Kauai. Monk seal sightings on Kauai have increased in recent years, with frequent haul-out behavior observed at the south shore. This contrasts with only one monk seal observation off the north shore of Kauai during 1994 shore-based Marine

Mammal Research Program surveys (Smultea *et al.* 1994). Despite these haul-out and pupping observations, little is known about the distribution and movement patterns of monk seals at sea (Gilmartin 1983; U.S. Dept. of Commerce 1986).

The green fur that is apparent on many seals results from an alga that grows on the fur, which attests to a prolonged period at sea (Kenyon and Rice 1959). Their distribution, destinations, routes, food sources, and causes of the movements when not traveling between islands are not well known (Johnson 1979), but recent tagging studies have shown individuals sometimes travel between the breeding populations in the northwest Hawaiian Islands. At the breeding islands, monk seals feed on octopus, spiny lobster, eels, bottom fish, and reef fish (Rice 1960; Gilmartin 1983).

Population status and trends

Consistent declines in the monk seal population trends have been recorded since surveys commenced in the late 1950s. Counts of Hawaiian monk seals made since the late 1950s and 1980s at the atolls, islands, and reefs where they haul out on the northwest Hawaiian Islands showed a 50% population decline (NMFS 1991). The total population for the five major breeding locations plus Necker Island for 1987 was estimated to be 1,718 seals including 202 pups of the year (Gilmartin 1988). This compares with 1,488 animals estimated for 1983 (Gerrodette 1985). In 1992 the Hawaiian monk seal population was estimated to be 1580 (SE=147) (Ragen 1993). The best estimate of total abundance for 1993 was 1,406 (SE=131, assuming a constant coefficient of variation). Thus, between 1958 and 1993, mean beach counts declined by 60 percent. For the years 1985 to 1993 the mean beach counts declined by approximately 5 percent per year. This downward trend is expected to continue, mainly due to poor pup and juvenile survival in recent years.

NMFS (2000) estimates the current monk seal population to be between 1,300 and 1,400 individuals. Data collected at five major haul-outs recorded a 23 percent decline in the number of births in 1990 from the average annual levels recorded between 1983 and 1989 (NMFS 1991).

Diving and social behavior

Limited data on the diving patterns of monk seals indicate that about half of the foraging activity of adult males occurs at depths shallower than 35 m (NMFS 1991). Recent time-depth recorder information from a tagged monk seals revealed that some monk seals sometimes dive to deeper than 300 m, with some dives recorded deeper than 500 m (Parrish *et al.*, 2000). Hawaiian monk seals can stay submerged for at least 20 minutes (Reeves *et al.* 1992).

Vocalizations and hearing

Studies on the vocal behavior of monk seals are limited. Job *et al.* (1995) found that female Hawaiian monk seals do not identify individual pups from by their vocalizations. Hawaiian monk seals have their most sensitive hearing at 12 to 28 kHz. Below 8 kHz, their hearing is less sensitive than other pinnipeds. Their sensitivity to high frequency sound drops off sharply above 30 kHz (Thomas *et al.* 1990, Richardson *et al.* 1995).

Impacts of human activity on this species

Populations of all monk seals have been reduced to either extinction (Caribbean monk seal), or near extinction (Mediterranean and Hawaiian monk seals). The main threats have been commercial and subsistence hunting, intentional harassment, competition with commercial fisheries, entanglement in fishing gear, habitat destruction on breeding beaches, pollution, and unintentional human disturbance (Kenyon 1981; Riedman 1990; Reeves *et al.* 1992).

Hawaiian monk seals have been protected from most adverse effects of human activities throughout most of their distribution, but the population is only recovering slowly (Reeves *et al.* 1992). There are no data on the potential effects of loud low frequency sound on monk seals.

Blue whale

Species description and distribution

Blue whales are the largest living mammal species. They may measure over 30 meters in length and weigh up to 160 metric tons. They are blue-gray in color with distinct gray and white mottling, while their ventral surface may be light pink in coloration. Their dorsal fin is relatively small. Like other baleen whales, they have fringed baleen plates instead of teeth, and ventral grooves which filter large quantities of water during feeding. Blue whales are found in all major oceans, including the continental shelf in coastal shelves and far offshore in pelagic environments of the North Pacific.

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 subpopulations 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 Commissions (IWC) Scientific Committee has formally recognized one blue whale population in the North Pacific (Donovan 1991), although there is increasing evidence that more than one population occurs in the Pacific Ocean (Gilpatrick *et al*. 1997, Barlow *et al*. 1995, Mizroch *et al*. 1984a, Ohsumi and Wada 1974). There have been no confirmed sightings or strandings of blue whales in the Hawaiian Islands area, but recordings of vocalizations (Thompson and Friedl 1982) suggest that blue whales are present within the U.S. Exclusive Economic Zone (U.S. EEZ) around Hawaii. The recordings showed peaks in summer and winter. Blue whale calls have also been recorded in Alaskan waters from 1995 to 1999 in every season although the whales have not been seen. Most of these calls occurred in fall and winter in the Gulf of Alaska suggesting that some blue whales remain in the area (as opposed to migrating through it).

Blue whales are found in the Atlantic Ocean from the Arctic to at least the mid-latitude waters of the North Atlantic with occasional occurrences in the U.S. EEZ (CeTAP 1982, Wenzel *et al.* 1988, Yochem and Leatherwood 1985, Gagnon and Clark 1993). Blue whales are most frequently sighted off eastern Canada. During winter, they are found in the waters off Newfoundland. In summer, they are found in Davis Strait (Mansfield 1985), in 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). Blue whales have been sighted off the Azores Islands, but Reiner *et al*. (1993) do not consider them common in that area.

Blue whales are found along the coastal shelves of North America and South America (Rice1974; Donovan1984; Clarke1980) in the North Pacific Ocean. The International Whaling Commission's (IWC) Scientific Committee has formally recognized one blue whale stock in the North Pacific (Donovan 1991). However, there is mounting evidence that more than one stock exists within this ocean basin (Gilpatrick *et al*. 1997; Barlow *et al*. 1994b; Braham 1991; Mizroch *et al*. 1984a; Ohsumi and Wada 1974). One such tentative stock designation is for concentrations of blue whales found during winter off Baja California and in the Gulf of California. Photo-identification studies have shown that individuals from these southern concentrations travel in the summer and fall to waters off California (Sears *et al*.1987; Barlow *et al*.1997; Calambokidis *et al*. 1990). Preliminary studies of these California/Mexico whales, based on length data from whaling records and aerial photogrammetry, have shown that they are morphologically distinct from blue whales of the western and central North Pacific (Gilpatrick *et al*. 1997).

In Hawaiian waters, acoustic monitoring has resulted in blue whales being recorded off Oahu and the Midway Islands (Barlow *et al*. 1994b; Northrop *et al*. 1971; Thompson and Friedl 1982), although sightings or strandings in Hawaiian waters have not been reported. Nishiwaki (1966) notes the occurrence of blue whales among the Aleutian Islands and in the Gulf of Alaska. However, as of 1987, there have been no blue whale sightings in these waters (Leatherwood *et al*. 1982; Stewart *et al*. 1987; Forney and Brownell 1996). No distributional information exists for the western region of the North Pacific.

The Costa Rica Dome is a stationary eddy in the eastern tropical Pacific that appears to be important for blue whales based on the high levels of prey (euphausiids) available in the Dome, the aggregation of juvenile whales, and the number of blue whales that appear to reside there (Reilly and Thayer 1990). Blue whales have been sighted in the Dome area throughout all seasons, but there is a peak in occurrence from June through November.

In the Atlantic Ocean, blue whales are found from the Arctic to least the mid-latitude waters of the North Atlantic (CeTAP 1982, Wenzel *et al*.1988, Yochem and Leatherwood 1985, Gagnon and Clark 1993). The IWC treats these whales as one stock (Donovan1991).

Sightings of blue whales occur most frequently off eastern Canada. During winter, they are found in the waters off Newfoundland. In summer, they are found in Davis Strait (Mansfield 1985), in 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 1992, the U.S. Navy conducted an extensive acoustic survey of the North Atlantic using the Integrated Underwater Surveillance System's (IUSS) fixed acoustic array system (Clark 1995). This study gave researchers insight into the seasonality of baleen whale vocalizations (Clark *et al*. 1993). 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).

Blue whales have 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.

Blue whales in the Southern Hemisphere are assigned to six stock areas designated by the IWC (Donovan 1991). These areas presumably follow the feeding distribution of the whales, although current distributional information on blue whales is still scarce. Historical catch records show that the true blue whale and the pygmy blue whale (*B. m. brevicada*) may be geographically segregated (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 summer, the true blue whale is found close to the ice-edge (south of 58° S) with concentrations between 60°-80° E and 66°-70° S (Kasamatsu *et al*. 1996). No new information on wintering areas has been reported since Braham's 1991 status review, so there is still no reproductive data to validate the stock designations in the Southern Hemisphere.

Life history information

Blue whale reproductive activities occur primarily in winter (see Yochem and Leatherwood 1985). Gestation takes 10−12 months, followed by a nursing period that continues for about 6-7 months. They reach sexual maturity at about 5 years of age (see Yochem and Leatherwood 1985). The age distribution of blue whales is unknown and little information exists on natural sources of mortality (such as disease) and mortality rates. Killer whales are known to attack blue whales, but the rate of these attacks or their effect on blue whale populations is unknown.

Important foraging areas include the edges of continental shelves and ice edges in polar regions (Yochem and Leatherwood 1985; Reilly and Thayer 1990). Data indicate that some summer feeding takes place at low latitudes in “upwelling-modified” waters (Reilly and Thayer 1990), and that some whales remain year-round at either low or high latitudes (Yochem and Leatherwood 1985; Clark and Charif 1998). The species *Thysanoëssa inermi*s, *T. longipe*s, *T. raschi*i, and *Nematoscelis megalops* have been listed as prey of blue whales in the North Pacific (Kawamura 1980; Yochem and Leatherwood 1985).

Although some stomachs of blue whales have been found to contain a mixture of euphausiids and copepods or amphipods (Nemoto 1957; Nemoto and Kawamura 1977), it is likely that the copepods and amphipods were consumed adventitiously or incidentally. One exception to their near- total dependence on euphausiid prey is that blue whales have been observed feeding on pelagic red crabs, *Pleuroncodes planipe*s, off Baja California (Rice 1974, 1986a), although these observations have not been confirmed by subsequent observations or other analyses (e.g., fecal analysis). Reports that blue whales feed on small, schooling fish and squid in the western Pacific (Mizue 1951; Sleptsov 1955) have been interpreted as suggesting that the zooplankton blue whales prefer are less available there (Nemoto 1957). Between February and April, blue whales in the Gulf of California, Mexico, have been observed feeding on euphausiid surface swarms (Sears 1990) consisting mainly of *Nyctiphanes simplex* engaged in reproductive activities (Gendron 1990, 1992). Sears (1990) regarded *Nyctiphanes simplex* as the principal prey of blue whales in the region, and results from recent fecal analyses confirmed this assertion (Gendron and Del Angel-Rodriguez 1997). However, this phenomenon appears to be strongly influenced by the occurrence of El Niño Southern Oscillation (ENSO) events (Gendron and Sears 1993). Other baleen whales whose range overlaps with the range of blue whales could potentially compete with blue whales for food (Nemoto 1970). Nevertheless, there is no evidence of competition among these whales and the highly migratory behavior of blue whales may help them avoid competition with other baleen whales (Clapham and Brownell 1996).

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 rorquals such as fin whales are regularly reported (Schoenherr 1991; Fiedler *et al.* 1998; Croll and Tershy pers. obs.). Little is known of the mating behavior of blue whales.

Vocalizations and hearing

Known vocalizations of 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; Ljungblad *et al.* in press). 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 (Clark pers. obs., McDonald pers. comm.). The seasonality and structure of long patterned sounds suggest that these sounds are male displays for attracting females and/or competing with other males. 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).

The function of vocalizations produced by blue whales is unknown. Hypothesized functions include: 1) maintenance of inter-individual distance, 2) species and individual recognition, 3) contextual information transmission (e.g., feeding, alarm, courtship), 4) maintenance of social organization (e.g., contact calls between females and offspring), 5) location of topographic features, and 6) location of prey resources (review by Thompson *et al.* 1979). 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 blue whale auditory apparatus, Ketten (1997) hypothesized that blue whales have acute infrasonic hearing. No studies have directly measured the sound sensitivity of blue whales.

Listing status

Blue whales have been listed as endangered under the ESA since 1973. They are also protected by the Convention on International Trade in Endangered Species of wild flora and fauna and the MMPA. The North Pacific stock is also listed as “low risk, conservation dependent” under the IUCN Red List of Threatened Animals (Baillie and Groombridge 1996). Critical habitat has not been designated for blue whales.

Population status and trends

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) which is a fraction of pre-whaling populations estimates of 200,000 animals. 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 at between 1,400 to 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.

The size of the blue whale population in the north Atlantic is also uncertain. The population has been estimated from a few hundred individuals (Allen 1970; Mitchell 1974) to 1,000 to 2,000 individuals (Sigurjónsson 1995). Gambell (1976) estimated there were between 1,100 to 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. Sigurjónsson and Gunnlaugson (1990) concluded that the blue whale population had been increasing since the late 1950s; from 1979 to 1988, they concluded that the blue whale population was increasing at an annual rate of about 5 percent.

Estimates of the Southern Hemisphere population range from 5,000 to 6,000 (review by Yochem and Leatherwood 1985) with an average rate of increase of 4 to 5 percent per year, but Butterworth *et al.* (1993) 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)

Impacts of human activity on this species

From 1889 to 1965 approximately 5,761 blue whales were taken from the North Pacific Ocean (NMFS 1998). Evidence of a population decline can be seen in the catch data from Japan. In 1912, 236 blue whales were caught, 58 whales in 1913, 123 whales in 1914, and from 1915 to 1965, the catch numbers declined continuously (Mizroch *et al.* 1984a). In the eastern North Pacific, 239 blue whales were taken off the California coast in 1926. And, in the late 1950s and early 1960s, Japan caught 70 blue whales per year off the Aleutian Islands (Mizroch *et al.* 1984a). The IWC banned commercial whaling in the North Pacific in 1966, since that time there have been no reported blue whale takes. Nevertheless, Soviet whaling probably continued after the ban so Soviet catch reports under-represent the number of blue whales killed by whalers (as cited in Forney and Brownell 1996). Surveys conducted in these former-whaling areas in the

1980s and 1990s failed to find any blue whales (Forney and Brownell 1996).

There are no reports of fisheries-related mortality or serious injury in any of the blue whale stocks. Blue whale interaction with fisheries may go undetected because the whales are not observed after they swim away with a portion of the net. However, fishers report that large blue and fin whales usually swim through their nets without entangling and with very little damage to the net (Barlow *et al.* 1997).

In 1980, 1986, 1987, and 1993, ship strikes have been implicated in the deaths of blue whales off California (Barlow *et al.* 1997). In addition, several photo-identified blue whales from California waters were observed with large scars on their dorsal areas that may have been caused by ship strikes. 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). The number of blue whales struck and killed by ships is unknown because the whales do not always strand or examinations of blue whales that have stranded did not identify the traumas that could have been caused by ship collisions. In the California/Mexico stock, annual incidental mortality due to ship strikes averaged 0.2 whales during 1991−1995 (Barlow *et al.* 1997), but we cannot determine if this reflects the actual number of blue whales struck and killed by ships.

Blue whales do not appear to be disturbed by noise from seismic exploration. When noise pulses from air guns were produced off Oregon, blue whales continued vocalizing at the same rate as before the pulses, suggesting that at least their vocalization behavior was undisturbed by the noise (McDonald *et al*. 1993).

Bowhead Whale

Species description and distribution

Bowhead whales are the second largest whales in the world, second only to the blue whale (*Balaenoptera musculus*) . The name "bowhead" comes from their bow-shaped mouth. The lower jaw makes a U-shape around the upper jaw. This lower jaw is usually marked with white spots, contrasting with the rest of the whale's black body (Nowak 1999). Baleen in the bowhead whale's mouth is the largest of any cetacean with 300 baleen plates measuring 300-450 centimeters in vertical length.

Bowhead whales were historically found in all arctic waters of the northern hemisphere. The International Whaling Commission currently recognizes five populations of bowhead whales: Sea of Okhotsk which occurs in the north Pacific Ocean off the western coast of Siberia near the

Kamchatka Peninsula; Bering Sea; Hudson Bay; Davis Strait, which is found in Davis Strait, Baffin Bay, and along the Canadian Arctic Archipelago; and Spitsbergen, which is found in the North Atlantic Ocean east of Greenland in the Greenland, Kara, and Barents Seas (IWC 1992). A separate Bering Sea population may have become extinct as a result of whaling activities, except for the component that migrated to the Beaufort Sea.

The Bering Sea population, which is also known as the western Arctic or Bering-Chukchi-Beaufort population, has been studied more than any other bowhead whale population. This population winters in the central and western Bering Sea (November to April) and migrates north and east through the eastern Chukchi Sea to the Beaufort Sea along the coast of Alaska and northwestern Canada (Brueggeman 1982, Braham *et al.* 1984). From June through September, these bowhead whales remain on foraging grounds in the eastern Beaufort Sea before migrating back to their wintering grounds in the Bering Sea (Hazard and Cubbage 1982; Richardson *et al.* 1987).

Bowhead whales in the western North Atlantic are currently segregated into two populations: the Davis Strait population occupies the Davis Strait, Baffin Bay, and the Canadian Arctic Archipelago while the Hudson Bay population occupies Hudson Strait, Hudson Bay, and Foxe Basin (Moore and Reeves 1993).

The Spitsbergen bowhead whale population, which is also known as the Greenland whale, bowhead whales in the eastern North Atlantic have been observed in the waters north of Iceland and as far east as the Laptev Sea. Shelden and Rugh (1995) reported sightings along the coastline of Greenland, in the waters near Spitsbergen Island, off North Cape in northern Norway, in the waters of Zemlya Frantsa-losifa (Franz Josef Land), near Novaya Zemlya, and near Severnaya Zemlya.

Life history information

Female bowhead whales are sexually mature when they reach lengths between 12.2 and 14.2 m, while males are sexually mature when they reach lengths between 12.6 and 13.6 m (George *et al.* 1998). Their gestation period has been estimated at 13 to 14 months (Nerini *et al.* 1984). Calving intervals range between 3 and 4 years (Nerini *et al.* 1984). Bowhead whales are assumed to mate in late winter and spring (Koski *et al.* 1993), perhaps continuing through the spring migration (Koski *et al.* 1993). Most calves are born from April through early June during the spring migration, with a few calves born as early as March and as late as August (Koski *et al.* 1993). Calves are about 13 to 15 ft (4 to 4.5 m) at birth and reach 42 to 66 ft (13 to 20 m) as adults. Females produce a single calf, probably every 3 to 4 years (Koski *et al.* 1993).

Bowhead whales are filter-feeders, sieving prey from the water by means of baleen fibers in their mouth. They feed almost exclusively on zooplankton from the water column, with primary prey consisting of copepods (54%) and euphausiids (42%), as indicated from stomach analyses of whales taken in the Alaskan Beaufort Sea (Lowry 1993). Other prey include mysids, hyperiid and gammarid amphipods, other pelagic invertebrates, and small fish. Bowheads feed heavily in the Canadian Beaufort Sea and Amundsen Gulf area during summer, and feeding is also known to occur during the fall migration through the Alaskan Beaufort Sea (Alaska Clean Seas 1983, Ljungblad *et al.* 1987, Lowry 1993). In surveys conducted from 1979 through 1987, concentrations of feeding bowheads were observed east of Point Barrow and just north of Harrison Bay during some years (Ljungblad *et al.* 1987). The majority of whales harvested during fall at Barrow have had food in their stomachs. Observations of feeding bowheads in 1998 found the whales feed primarily along the Alaskan coast near Kaktovik, but that feeding locations vary among years (Richardson and Thomson 1999).

Diving and social behavior

Bowhead whales can dive as deep as 455 m (Krutzikovsky and Mate 1993). Diving characteristics of whales from the Davis Strait population include time at surface, 1.5 minutes per surfacing with an average of 6 blows per surfacing and mean dive times of 9.3 minutes (Würsig and Clark 1993). Diving characteristics from the Bering Sea population include an average of 1.7 minutes per surfacing, 6.5 blows per surfacing, and a mean dive time of 19.2 minutes (Würsig and Clark 1993).

Vocalizations and hearing

Generally, the vocalizations of bowhead whales are low, less than 400 Hz frequency-modulated calls; however, their call repertoire also includes a rich assortment of amplitude-modulated and pulsed calls with energy up to at least 5 kHz (Wursig and Clark 1993). Calls and songs have been suggested to be associated with different contexts and whale behavior. Observations have been made that support the theory that calls are used to maintain social cohesion of groups. For instance, loud frequency-modulated calls were heard as a mother and a calf rejoined after becoming separated during summer feeding (Wursig and Clark 1993). Once the two were together again, calling stopped (Wursig and Clark 1993). During spring migration off Point Barrow, there have been several instances when individual whales repeatedly produced calls with similar acoustic characteristics (Clark *et al.* 1987). Bowhead whales have been noted to produce signature calls lasting for 3 to 5 minutes each and continuing up to 5 hours (Wursig and Clark 1993). Different whales produce signature calls as they counter call with other members of their herd. It has been suggested that calling among bowhead whales may aid in migration of the herd and that the surface reverberation of the sound off the ice may allow these whales to discriminate among areas through which they can and cannot migrate (Ellison *et al.* 1987, Wursig and Clark 1993).

It has been speculated that bowheads are able to locate leads and open water along the marginal ice zone in winter by using acoustics (Moore and Reeves 1993). Although bowheads are morphologically adapted to their ice-dominated environment and can break holes in the ice to breathe, they may use vocalization to assess ice conditions in their path. For example, the intensity of reflected calls is as much as 20 decibels (dB) higher from ice floes with deeper keels than from relatively flat, thin ice (Ellison *et al.* 1987). Bowheads may use such differences in intensity of reflected calls to differentiate between deep keel ice floes and flat, thin ice.

Bowhead whales have extremely sensitive hearing. For example, they are capable of detecting sounds of icebreaker operations at a range of 31 miles (50 km) (Richardson 1996). It has been suggested that such sensitive hearing also allows whales to use reverberations from their low frequency calls to navigate under the pack ice and to locate open water polynyas where they surface (Ellison *et al.* 1987). Bowheads exhibit avoidance behavior at many manmade sounds, but there is still considerable debate regarding their range of sound detection (Richardson *et al.* 1995). It is well known among Inupiat hunters that bowhead whales are extremely sensitive to noise (H. Rexford in MMS 1979, R. Ahkivgak in NSB 1980, H. Ahsogeak in NSB 1980, T. Brower in NSB 1980, H. Brower in MMS 1990). Communications among whales during migration and in response to danger also has been observed to alter migration patterns (A. Brower in MMS 1986, T. Napageak in MMS 1995). Whaling crews have observed that disturbances to migration as a result of a strike are temporary (J.C. George in USACE 1996).

Listing status

Bowhead whales were listed as endangered species on June 2, 1970 (35 FR 8495). The bowhead gained further protection under the Convention on International Trade in Endangered Species of wild flora and fauna. Critical habitat has not been designated for bowhead whales.

Population status and trends

Before exploitation, the Sea of Okhotsk population may have numbered between 3,000 and 6,500 animals (Shelden and Rugh 1995); it is now estimated to number between 300 and 400 animals (although these population estimates are not reliable). Individuals from this population may have mixed with individuals from the Bering Sea population, although the available evidence indicates the two stocks are essentially separate (Moore and Reeves 1993).

The Bering Sea population of bowhead whales declined from an estimated population of 10,400 to 23,000 animals (Woodby and Botkin 1993); by 1910, this population had been reduced to a few thousand individuals. From 1978 to 1983, this population was estimated to have numbered between 3,500 to 5,300 animals based on shore-based visual surveys (Zeh *et al.* 1993). The IWC Scientific Committee now recognizes the current population estimate to be 7,992 whales (95% C.I.: 6,900-9,200) (IWC 1995). A refined and larger sample of acoustic data from 1993 has resulted in an estimate of 8,200 animals, and is considered a better estimate for this population (IWC 1996).

The Spitsbergen stock was reduced from 24,000 to a few "tens" of whales and has not recovered in the past 80 years. The Davis Strait and Hudson Bay stocks declined from about 12,300 whales to less than 450 currently, although significant whaling has not occurred in 80 years.

There are no reliable estimates of the size of the Hudson Bay population of bowhead whales, although Mitchell (1977) conservatively estimates it at 100 or less. More recently, this population has been estimated to number from 256 to 284 whales within Foxe Basin (Cosens *et al.* 1997).

The Davis Strait population is separated from the Bering Sea stock by the heavy ice found along the Northwest passage (Moore and Reeves 1993). The population was estimated to have originally numbered over 11,700 (Woodby and Botkin 1993) but was significantly reduced by commercial whaling between 1719 and 1915. The Davis Strait population is currently estimated to be 350 animals (Zeh *et al.* 1993) and recovery is described as “at best, exceedingly slow” (Davis and Koski 1980). Canadian Inuit have expressed an interest in resuming subsistence hunting of bowhead whales in Davis Strait, although the International Whaling Commission has not acted on this request.

The Spitsbergen population of bowhead whales was believed to have been the most numerous of the bowhead whale populations: before they were hunted by commercial whalers, the population was estimated to number about 24,000 animals (Woodby and Botkin 1993). Between 1940 and September 1990, 37 bowhead whale sightings have been recorded in this region, but some of these sightings were not unequivocally bowhead whales (Moore and Reeves 1993). With a population size numbering in the tens of animals, the Spitsbergen population of bowhead whales is now critically endangered (Shelden and Rugh 1995).

Impacts of human activity on this species

The Bering Sea population of bowhead whales is harvested by Inupiat in the Alaskan Beaufort, Bering, and Chukchi Seas. Since 1978, the IWC has imposed a quota on the number of bowheads landed, struck, or both by Alaskan natives. The IWC recently allocated the subsistence take of bowheads from the Alaska stock, establishing a 5-year block quota of 280 whales landed. For each of the years 1998-2002, the number of bowheads struck is not allowed to exceed 67 animals. In addition, the Russian Federation for the Natives of Chukotka has been granted an annual quota of five bowheads.

The total Alaskan subsistence harvest of bowheads between 1978 and 1991 ranged from 8 in 1982 to 30 in 1990, averaging 18 per year. From 1991 to 1995, a combined average of 19.4 bowhead whales per year were taken by the communities of Barrow, Nuiqsut, and Kaktovik (MMS 1996). The combined spring and fall harvest for 1998 was 41 whales landed and 12 struck and lost. In addition to the subsistence harvest, other man-induced impacts may contribute to morbidity and mortality. Commercial fishing occurs in the Bering Sea and elsewhere within the range of this stock. Interaction with fishing gear is rare, however whales with ropes caught in their baleen and with scarring caused by rope entanglement have been reported (Philo *et al.* 1993, NMML unpubl. data).

George *et al.* (1994) report three documented ship strike injuries observed among 236 bowheads taken in subsistence hunts. Man-made noise in arctic marine environments is increasing as the region becomes industrialized; these activities may adversely affect bowhead whales but the significance of those effects, if any, remain unknown (Richardson and Malme 1993, Richardson *et al.* 1995).

Shelden and Rugh (1995) suggested that the longevity and low fecundity rates of bowhead whales may be important factors in the slow recovery of bowhead whales since the termination of whaling.

Fin whale

Species description and distribution

Fin whales are distributed widely in the world's oceans. In the northern hemisphere, most migrate seasonally from high Arctic feeding areas in summer to low latitude breeding and calving areas in winter. Other groups may remain year-round in a particular area, depending on food supply. The IWC's Scientific Committee recognizes two management stocks in the North Pacific: (1) the east China Sea, and (2) the rest of the North Pacific (Donovan 1991). Mizroch *et al.* (1984b) suggested five possible stocks within the North Pacific based on histological 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 the Gulf of Alaska; and (5) Gulf of California (Rice 1974, Tershy *et al.* 1993). However, NMFS considers stock structure in the North Pacific to be equivocal, and recognizes three stocks: (1) Alaska (northeast Pacific), (2) California/Oregon/ Washington, and (3) Hawaii (Barlow *et al.* 1997, Hill and DeMaster 1998).

Fin whales were reported as occurring immediately offshore throughout the North Pacific from central Baja California to Japan and as far north as the Chukchi Sea (Rice 1974). Data indicate that some whales remain year-round at high latitudes (Clark and Charif 1998) and other areas such as the Gulf of California (J. Urban, UABCS, La Paz, BCS. Mexico, pers. comm.), migrating only short distances of 100-200 km (53.9-107.9 nm) (Agler *et al.* 1993). In the Gulf of Alaska, fin whales appear to congregate in the waters around Kodiak Island and south of Prince William Sound. In recent years, small numbers of fin whales have been observed south of the Aleutian Islands (Forney and Brownell 1996), in the Gulf of Alaska (including Shelikof Strait), and in the southeastern Bering Sea (Leatherwood *et al.* 1986). Fin whale concentrations in the northern areas of the North Pacific and Bering Sea generally form along frontal boundaries, or mixing zones between coastal and oceanic waters, which themselves correspond roughly to the 200-m isobath (which is the shelf edge; Nasu 1974).

Acoustic data collected from 1995 to 1999 from hydrophone arrays show fin whales vocalizing in Alaskan waters during all seasons, with a peak in occurrence in midwinter. Fin whales are rare in Hawaiian waters, but may occur within 200 nm (370 km) of Hawaii during winter months, when some of the fin whales disperse throughout the lowest latitudes of their distribution (Balcomb 1987). Acoustic recordings (Thompson and Friedl 1982) suggest that fin whales migrate into Hawaiian waters (the U.S. EEZ surrounding the Hawaiian Archipelago) primarily during the fall and winter. More recently, McDonald and Fox (1999) reported an average of 0.027 calling fin whales per 1000^2 km (grouped in 8 hour periods) based on passive acoustic recording within about 16 km of the north shore of Oahu. A single fin whale sighting occurred approximately 37 km (20 nm) north of Kauai in 1994 (Mobley *et al.* 1996). Two confirmed sightings and one stranding comprise the records for fin whales in this region (not including whaling records) (Nitta 1987).

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.

Life history information

Fin whales become sexually mature between six to ten years of age, depending on density-dependent factors (Gambell 1985). Reproductive activities for fin whales occur primarily in the winter. Gestation lasts about 12 months and nursing occurs for 6-11 months (Perry *et al.* 1999). The age distribution of fin whales in the North Pacific is unknown. Calving and mating occur in late fall and winter (Millais 1906; Mackintosh and Wheeler 1929; Nishiwaki 1952; Tomilin 1957). Specific breeding areas are unknown and mating is assumed to occur in pelagic waters, presumably some time during the winter when whales are in mid-latitudes. Fin whales commonly travel in herds ranging from 6 to12 individuals, to nearly 100 or more (Balcomb 1987).

Foraging areas tend to occur along continental shelves with productive upwellings or thermal fronts (Gaskin 1972; Sergeant 1977; Nature Conservancy Council 1979). Fin whales tend to avoid tropical and pack ice waters (Meredith and Campbell 1988), with the northern limit of their range set by ice and the southern limit by warm water of approximately 15°C (60°F) (Sergeant 1977). Fin whales in the North Pacific feed on euphausiids, calanoid copepods, and schooling fish such as herring, pollack, Atka mackerel, and capelin (Calkins 1986; Nemoto 1957, 1970; Kawamura 1982). Euphausiids may be preferred prey, and competition may occur with other baleen whales or other consumers of these prey types. Natural sources and rates of mortality are largely unknown, but Aguilar and Lockyer (1987) suggest annual natural mortality rates may range between 0.04 and 0.06 (based on studies of northeast Atlantic fin whales). 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 result in serious injury or death in very young and sick whales (Perry *et al.* 1999). NMFS has no records of fin whales being killed or injured by commercial fisheries operating in the North Pacific (Ferrero *et al.* 2000).

Diving and social behavior

Generally, fin whales make 5-20 shallow dives 13-20 seconds in duration followed by a deep dive of 1.5 to 15 minutes (Gambell 1985; Strong 1990; Croll and Tershy, pers. obs.). Croll and Tershy (pers. obs.) recorded dive depths of 100-200 m (330-660 ft), with maximum depths of 300 m (1,000 ft). Dive depths and duration were significantly shorter at night than during the day, presumably in response to the daily vertical migrations of prey schools. An estimate of dive depth based on the acoustical properties of received fin whale calls was 525 m (1,722 ft) (Charif *et al.*, submitted).

Fin whales are often found singly or in pairs, but also commonly form larger groupings greater than 3 individuals, particularly while feeding. Tershy *et al.* (1993) described group foraging behavior where 2-4 animals swam less than 50 m apart in an echelon formation and lunged synchronously, right side down. They found that group composition was not stable: membership and group size changed frequently during feeding events.

Vocalizations and hearing

Underwater sounds of the fin whale 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 pers. comm.; McDonald pers. comm.). Each pulse lasts on the order of one second and contains twenty cycles (Tyack 1999).

Particularly in the breeding season, fin whales produce 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 is geographic differences in the frequency, duration and repetition of the pulses (Thompson *et al.* 1992). As with other mysticete sounds, the function of vocalizations produced by fin whales is unknown. Hypothesized functions include: 1) maintenance of inter-individual distance, 2) species and individual recognition, 3) contextual information transmission (e.g. feeding, alarm, courtship), 4) maintenance of social organization (e.g. contact calls between females and offspring), 5) location of topographic features, and 6) location of prey resources (review by Thompson *et al.* 1992). 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).

A description of the anatomy of the ear for cetaceans is provided in the description of the blue whale above. No studies have directly measured the sound sensitivity of fin whales. Presumably fin whales are able to receive sound signals of the same frequency they are producing. In a study of the morphology of the mysticete auditory apparatus, Ketten (1997) hypothesized that large mysticetes have acute infrasonic hearing.

Listing status

In the North Pacific, the IWC began management of commercial whaling for fin whales in 1969; fin whales were fully protected from commercial whaling in 1976 (Allen 1980). Fin whales are listed as endangered under the ESA. They are also protected by the Convention on International Trade in Endangered Species of wild flora and fauna and the MMPA. Fin whales are listed as endangered on the IUCN Red List of Threatened Animals (Baillie and Groombridge 1996). Critical habitat has not been designated for fin whales.

Population status and trends

Prior to exploitation by whaling vessels, the North Pacific population consisted of an estimated 42,000-45,000 fin whales (Ohsumi and Wada 1974). Between 1914 and 1975, over 26,040 fin whales were harvested throughout the North Pacific (in Perry *et al.* 1999). Catches in the North Pacific and Bering Sea ranged from 1,000 to 1,500 fin whales annually during the 1950's and 1960's. However, not all Soviet catches were reported (cited in Ferrero *et al.* 2000). In the early 1970s, the entire North Pacific population had been reduced to between 13,620 and 18,630 fin whales (Ohsumi and Wada 1974). During the early 1970s, 8,520-10,970 fin whales were surveyed in the eastern half of the North Pacific (Braham 1991). If these historic estimates are statistically reliable, the population size of fin whales has not increased significantly over the past 20 years despite an international ban on whaling in the North Pacific. The current status and trend of the fin whale population in the North Pacific is largely unknown. Based on the available information, it is feasible that the North Pacific population as a whole has failed to increase significantly over the past 20 years, despite an international ban on whaling in the North Pacific. The only contrary evidence comes from investigators conducting seabird surveys around the Pribilof Islands in 1975-1978 and 1987-1989. These investigators observed more fin whales in the second survey and suggested they were more abundant in the survey area (Baretta and Hunt 1994). A survey for whales in the central Bering Sea in 1999 tentatively estimated the fin whale population was about 4,951 animals (95% C.I.: 2,833-8,653).

Hain *et al*. (1992) estimated there were about 5,000 fin whales in the western North Atlantic Ocean based on a 1978-1982 survey. The current best estimate places this population at about 2,200 animals with a minimum estimate of about 1,800 (Waring *et al*. 1999). The East Greenland/Iceland fin whale population has been estimated at 10,000 animals (95 % CI 7,600-14,200), based on surveys conducted in 1987 and 1989 (Buckland *et al*. 1992a). The number of eastern Atlantic fin whales, which includes the British Isles-Spain-Portugal stock, is estimated at 17,000 animals (95% CI 10,400-28,900; Buckland *et al*. 1992). Reiner *et al*. (1993) associate fin whales with waters off the Azores, although we cannot determine which population those fin whales might represent.

Fin whales are the most common, large cetacean in the Mediterranean Sea and are frequently reported in the western Mediterranean (Gannier 1998). Forcada (1996) estimated the fin whale population in the western Mediterranean at 3,583 individuals (SE: 967 95% CI: 2,130-6,027). During the summer, fin whales seem to congregate in the highly productive waters of the north-western Mediterranean basin, which includes the Ligurian Sea.

Impacts of human activity on this species

As early as the mid-seventeenth century, the Japanese were capturing fin, blue, 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. The North Pacific and Antarctic whaling operations soon added this 'modern' equipment to their arsenal. After blue whales were depleted in most areas, the smaller fin whale became the focus of whaling operations and more than 700,000 fin whales were landed in the twentieth century. In the North Pacific, there are no reports of fin whale deaths caused by fishery-related activities (Hill *et al.* 1997), although conflicts between fin whales and drift gillnet fisheries may occur (Barlow *et al.* 1997). Because of their size, strength, and distribution, it is difficult to assess potential interactions between fin whales and fisheries; for example, fishermen have reported that large blue and fin whales usually swim through their nets without entangling and with very little damage to the net (Barlow *et al.* 1997).

Gray whale [Western Pacific Population]

Species description and distribution

The western Pacific population of gray whales is distributed between the west central Sea of Okhotsk (from summer to fall) and the South China Sea (winter). The IUCN lists this gray whale population as critically endangered. Gray whales are the most coastal of the baleen whales and the western Pacific population is no exception: whales in this population generally forage in shallow, nearshore waters throughout the year except when crossing open-water passages.

Feeding grounds off northeastern Sakhalin Island, Russia appear to be important for this species during the summer. Their migratory route from these feeding grounds to winter rearing areas include regions off the eastern shore of Sakhalin Island in the Okhotsk Sea (Berzin in press), along the eastern shore of mainland Russia near the La Perouse and Tatarskiy Straits (Berzin in press), and off the eastern shore of the Korean peninsula (Andrews 1914, Brownell and Chun 1977). Data derived from whaling records suggest that gray whale numbers off Korea peaked in two pulses, one between December and January (probably during the southward migration) and the other between March and April (during the northward migration; Andrews 1914, Kato and Kasuya in press). In addition, gray whale have been occasionally sighted off the Pacific coast of southern Japan between 1959-1997 (Kato and Tokuhiro 1997, Omura 1984).

Winter calving and mating areas for this population remain unknown. However, some investigators have suggested that the western population of gray whales calves off the southern end of the Korean Peninsula (Andrews 1914), although the analyses that led to this conclusion are being debated (Rice and Wolman 1971). Historical records indicate that the western population of gray whales occurred as far south as the Yellow Sea, East China Sea, and South China Sea (Henderson 1972, 1984, 1990; Wang 1978, 1984; Omura 1988, Kato and Kasuya in press), so calving may occur in these areas. In addition, Omura (1974) suggested that an alternative or additional calving and mating area was in the Seto Inland Sea off southern Japan, although this suggestion has not been supported by empirical observation.

Life history information

Gray whales become sexually mature between six to ten years of age, depending on density-dependent factors (Gambell 1985). Reproductive activities for gray whales occur primarily in the winter. Gestation lasts about 12 months and nursing occurs for 6-11 months (Perry *et al.* 1999). The age distribution of gray whales in the North Pacific is unknown. Calving and mating occur in late fall and winter (Millais 1906; Mackintosh and Wheeler 1929; Nishiwaki 1952; Tomilin 1957).

Rice and Wolman (1971) reported the mean age of sexual maturity for eastern gray whales to be eight years (range: 5 to 11). Reilly (1992) estimated adult survival to be 0.95. Although there are no data for first year survival in gray whales, researchers have argued that the first-year survival estimate for humpback whales, which is 0.875, is probably similar to that of gray whales (Barlow & Clapham 1997). Rice and Wolman (1971) concluded that 24 percent of the eastern gray whales in their study were sexually immature, and suggested that the total proportion of immature animals was probably 44.0 to 61.0 percent. Females gray whales give birth every second year. Peak mating occurs in late November and early December.

Diving and social behavior

Gray whales feed primarily on benthic amphipods (Rice and Wolman 1971) by sucking sediment filled with amphipods off the sea floor and expelling the sediment and water through their baleen plates (Oliver *et al*. 1984; Nerini 1984; Guerrero 1985). In addition, gray whales will feed on other invertebrates that include mysiids, euphausiids, and pelagic crabs (Reeves and Mitchell 1988). Most feeding takes place on their northern feeding grounds, although Nerini (1984) reported evidence of extensive feeding during migration.

Gray whales are not deep divers. While foraging, they will generally remain in waters less than 80 m in depth. Their average dive times are between 4 and 5 minutes.

Vocalizations and hearing

Gray whales produce and probably hear low frequency sounds (Croll *et al*. 1999). Experimental playback studies have shown that gray whales avoid novel, low frequency sound sources, including sounds produced by SURTASS LFA, when received levels are about 160 dB (Croll *et al*. 1999). There are no data on the hearing range of gray whales.

Malme *et al*. (1983) reported that gray whales changed course or slowed down in response to playbacks of higher frequency components of a large helicopter. Malme *et al*. (1984) also reported that migrating gray whales slowed down or turned away from airgun pulses when received levels exceeded 160 dB re 1 μPa. Tyack and Clark (1998) reported that migrating whales avoided LFA playbacks with sources levels of 170 and 178 dB at several hundred meters. Clark *et al*. (1989) reported that gray whales, particularly females with young, reacted to small airplanes flying at elevations of 335 m.

Listing status

The North Atlantic population of gray whales became extinct as a result of whaling activity during the early 1900s. In the North Pacific, the IWC began management of commercial whaling for gray whales in 1969; gray whales were fully protected from commercial whaling in 1976 (Allen 1980). North Pacific gray whales were listed as endangered under the ESA in 1973. The eastern Pacific population of the gray whale was removed from the list of threatened and endangered species in 1994 (59 FR 31094). However, the western Pacific population of gray whales remains protected as an endangered species under the ESA. Critical habitat has not been designated for the western Pacific population of gray whales.

Gray whales are also protected by the Convention on International Trade in Endangered Species of wild flora and fauna and the MMPA. Gray whales were listed as critically endangered by IUCN in 2000 (Hilton-Taylor 2000).

Population status and trends

The western Pacific population of gray whales was thought to be extinct as recently as 1972 (Bowen 1974), but is known to survive today as a small remnant population (Berzin 1974, Brownell and Chun 1977, Weller *et al.* 1999). Aerial and ship-based sighting records in the Okhotsk Sea between 1979 and 1989 indicated that gray whales aggregated predominantly along the shallow-water shelf of northeastern Sakhalin Island, Russia, and were most common off the southern portion of Piltun Lagoon (Blokhin *et al.* 1985; Berzin 1988, 1990, 1991, in press; Blokhin 1996).

The current population size of western gray whales is unknown, but suspected to be approximately 100 (Weller *et al.* 1999). Two non-quantitative population estimates of 100 (Blokhin 1996) and 250 (Vladimirov 1994) have been reported in the Russian literature, but neither of these are valid. The estimate of 250 by Vladimirov (1994) was derived from Russian cetacean sighting records collected between 1979-1992. These records were collected from a variety of observation platforms, during different seasons, and employed mostly non-systematic sampling strategies. Although counts may be inflated by repeated observations of the same individuals, the highest count reported by Vladimirov (1994) during any sampling period was 34 gray whales observed off northeastern Sakhalin Island in 1989. The foundation for how this count of 34 was extrapolated to a population estimate of 250 is not described or based on any apparent quantitative calculation. The estimate of 100 by Blokhin (1996) was based on eight shore counts and one helicopter survey conducted between July-August of 1995 along the northeastern Sakhalin Island coast. The highest number of whales counted on any one day during this period was 42. As was true for the estimate of 250 by Vladimirov (1994), the logic for how Blokhin (1996) took the highest count of 42 and extrapolated to a population estimate of 100 is not described. Both of these estimates should be considered unreliable and not used as the basis for understanding the current population status of western gray whales.

Recent photo-identification studies (1994-1995 and 1997-1999) on the primary feeding grounds off northeastern Sakhalin Island have identified a total of 88 individual whales (Weller *et al.* 2000). These photo-identification data indicate high levels of annual return and pronounced seasonal site fidelity. While new individuals continue to be identified annually, the rate at which this is occurring is near the asymptote. Only 20 previously unidentified whales (excluding calves) were photographed during 91 days of effort between 1998-1999. This finding suggests that a majority of the population had been identified between 1994-1997 (Weller *et al*. 2000). Between 1995-1999, 11 reproductive females and their 13 calves were observed (Brownell *et al*. 1997, Weller *et al*. 1999, 2000). Two calves were observed in 1995, 1997, and 1999, and seven calves were sighted in 1998. Crude birth rates ranged between a low of 2.8% in 1999 to a high of 13.0% in 1998. Of the 11 calves identified between 1995-1998, seven (63.3%) have not been resighted on the Sakhalin feeding grounds subsequent to their birth year.

Using the minimum estimate of 88 western gray whales (Weller *et al*. 1999, 2000) and using the proportion immature from Rice and Wolman (1971), Brownell *et al*. (2000) estimated the number of mature whales in the western subpopulation at between 34 and 49. After assuming that all males in the population reproduced but only 70% of the females reproduced, Brownell *et al*. (2000) estimated that 85% of the sexually mature animals are capable of reproduction. With this assumption, the concluded that the western population of gray whales may only contain 50 breeding adults, making them one of the most critically-endangered cetaceans in the world.

Impacts of human activity on this species

The decline in the western Pacific gray whale population can be largely attributed to modern commercial whaling off Russia, Korea, and Japan between the 1890s and 1960s. This population has been legally protected under the International Convention for the Regulation of Whaling since 1946. The Republic of Korea and China, however, did not join the International Whaling Commission (IWC) until 1978 and 1980, respectively. Prior to their IWC membership, at least 67 gray whales were killed between 1948-1966 off the Republic of Korea, and the absence of catch reports from 1967 to 1980 does not necessarily indicate the absence of gray whale harvests by either of these countries during that fourteen year period (Brownell and Chun 1977). The Democratic People's Republic of Korea, recognized for its long-term involvement in coastal and pelagic whaling operations, is not currently a member of the IWC and nothing is known about this country's whaling activities over the past fifty years.

Current threats to the western gray whale population include continued mortality from an undetermined level of hunting (Brownell 1999, Brownell and Kasuya 1999), and incidental catches in the extensive coastal net fisheries off southern China (Zhou and Wang 1994). The substantial nearshore industrialization and shipping congestion throughout the migratory corridor(s) of this population also represent potential threats by increasing the likelihood of exposure to chemical pollution and ship strikes. Present and planned large-scale offshore gas and oil development in the South China Sea, and within 20 km of the only known feeding ground for western gray whales off northeast Sakhalin Island in the Okhotsk Sea, is of particular concern (Brownell *et al.* 1997, Würsig *et al.* 1999, 2000, Weller *et al*. 2000). Activities related to oil and gas exploration, including high-intensity geophysical seismic surveying, drilling operations, increased ship and air traffic, and oil spills, all pose potential threats to gray whales. Disturbance from underwater industrial noise may displace whales from critical feeding, migratory, and breeding habitat (Bryant *et al*. 1984, Richardson *et al*. 1995). Physical habitat damage from drilling and dredging operations, combined with possible impacts of oil and chemical spills on benthic prey communities also warrants concern.

Despite international agreements that prohibit harvests of these whales, at least one western gray whale was illegally killed off Hokkaido, Japan, in 1996 (Baker *et al*. 2002, Brownell 1999, Brownell and Kasuya 1999). Baker *et al*. (2002) report the sale of meat from seven gray whales, whose genetics apparently match the published sequence from Washington State, in Japan in 1999. Based on the results of their investigations, Baker *et al*. (2002) suggested that illegal hunting along the coast of Japan could be one of the factors inhibiting the recovery of this critically endangered population.

Humpback whale

Species description and distribution

NMFS recognizes four stocks of humpback whales in the North Pacific basin, 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).

Humpback whales typically migrate between tropical/sub-tropical and temperate/polar latitudes. Humpback whales feed on krill and small schooling fish on their summer grounds. The whales occupy tropical areas during winter months when they are breeding and calving, and polar areas during the spring, summer, and fall, when they are feeding, primarily on small schooling fish and krill (Caldwell and Caldwell 1983). It is believed that minimal feeding occurs in wintering grounds, such as the Hawaiian Islands (Balcomb 1987; Salden 1987). Humpback whales summer throughout the central and western portions of the Gulf of Alaska, including Prince William Sound, around Kodiak Island (including Shelikof Strait and the Barren Islands), and along the southern coastline of the Alaska Peninsula. The few sightings of humpback whales in offshore waters of the central Gulf of Alaska are usually attributed to animals migrating into coastal waters (Morris *et al.* 1983), although use of offshore banks for feeding is also suggested. The continental shelf of the Aleutian Islands and Alaska Peninsula were once considered the center of the North Pacific humpback whale population (Berzin and Rovnin 1966; Nishiwaki 1966). The northern Bering Sea, Bering Strait, and the southern Chukchi Sea along the Chukchi Peninsula appear to form the northern extreme of the humpback whale's range (Nikulin 1946, Berzin and Rovnin 1966).

Humpback whales occur off all eight Hawaiian Islands, but particularly within the shallow waters of the "four-island" region (Kaho'olawe, Molokai, Lanai, Maui), the northwestern coast of the Big Island, and the waters around Niihau, Kauai and Oahu (Wolman and Jurasz 1977; Herman *et al.* 1980; Baker and Herman 1981). The whales are generally found in shallow water shoreward of the 182 m (600-ft) depth contour (Herman and Antinoja 1977), although Frankel *et al.* (1989) reported some vocalizing individuals up to 20 km (10.8 nm) off South Kohala on the west coast of the Big Island, over bottom depths of 1400 m (4593 ft). Cow and calf pairs appear to prefer very shallow water less than 18 m (60 ft) (Glockner and Venus 1983). At Kuili off the Big Island, Smultea (1989) found significantly more cow/calf pairs in water <55 m (180.5 ft) deep. Some results suggest that habitat use patterns of nearshore waters by females and calves near Maui may have changed (decreased), potentially due to increasing vessel and other human activities (Salden 1988; Glockner-Ferrari and Ferrari 1990).

In the Atlantic Ocean, humpback whales feed in the northwestern Atlantic during the summer months and migrate to calving and mating areas in the Caribbean. Six separate feeding areas are utilized in northern waters after their return. This area will not be affected because it is within the biologically important area defined by the 200-m (656-ft) isobath on the North American east coast. Humpback whales also use the mid-Atlantic as a migratory pathway and apparently as a feeding area, at least for juveniles. Since 1989, observations of juvenile humpbacks in that area have been increasing during the winter months, peaking January through March (Swingle *et al.* 1993). Biologists theorize that non-reproductive animals may be establishing a winter feeding range in the Mid-Atlantic since they are not participating in reproductive behavior in the Caribbean. It is assumed that humpback whales are more widely distributed in the action area than right whales. They feed on a number of species of small schooling fishes, particularly sand lance and Atlantic herring, by targeting fish schools and filtering large amounts of water for the associated prey. Humpback whales have also been observed feeding on krill.

Life history information

Humpback whale reproductive activities occur primarily in winter. They become sexually mature at age four to six. Annual pregnancy rates have been estimated at about 0.40-0.42 (NMFS unpublished and Nishiwaki 1959). Cows will nurse their calves for up to 12 months. The age distribution of the humpback whale population is unknown, but the portion of calves in various populations has been estimated at about 4−12% (Chittleborough 1965, Whitehead 1982, Bauer 1986, Herman *et al.* 1980, and Clapham and Mayo 1987).

The information available does not identify natural causes of death among humpback whales or their number and frequency over time, but potential causes of natural mortality are believed to include parasites, disease, predation (killer whales, false killer whales, and sharks), biotoxins, and entrapment in ice.

Humpback whales exhibit a wide range of foraging behaviors, and feed on a range of prey types including small schooling fishes, euphausiids, and other large zooplankton. Fish prey in the North Pacific include herring, anchovy, capelin, pollack, Atka mackerel, eulachon, sand lance, pollack, Pacific cod, saffron cod, arctic cod, juvenile salmon, and rockfish. In the waters west of the Attu Islands and south of Amchitka Island, Atka mackerel were preferred prey of humpback whales (Nemoto 1957). Invertebrate prey include euphausiids, mysids, amphipods, shrimps, and copepods.

Diving and social behavior

In Hawaiian waters, humpback whales remain is almost exclusively within the 1820 m isobath and usually within 182 m. 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 unpubl. manus.). In southeast Alaska average dive times were 2.8 min for feeding whales, 3.0min 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.

Clapham (1986) reviewed the social behavior of humpback whales. They form small unstable 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 grounds (Clapham 1994, 1996), and on wintering ground (Tyack 1981). On the breeding grounds 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

Humpbacks 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 six to 12 miles (10 to 20 km). Animals in mating groups produce a variety of sounds (Tyack 1981; Tyack and Whitehead 1983, Silber 1986). Sounds are produced less frequently on the summer feeding grounds. 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, which 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 μPa-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).

A general description of the anatomy of the ear for cetaceans is provided in the description of the blue whale above. Humpback whales respond to low frequency sound. 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 102dB (Frankel *et al.* 1995). Humpback whales apparently reacted to 3.1 – 3.6 kHz sonar by changing behavior (Maybaum 1990 1993). Malme *et al.* (1985) found no clear response to playbacks of drill ship and oil production platform noises at received levels up to 116dB 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 likely 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 playback of 60 – 90 Hz bounds with a received level of up to 190 dB. While these studies have shown short-term behavioral reactions to boat traffic and playbacks of industrial noise, the potential for habituation, and thus the long-term effects of these disturbances are not known.

Listing status

The IWC first protected humpback whales in the North Pacific in 1965. Humpback whales were listed as endangered under the ESA in 1973. 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 the species.

Population status and trends

An estimated 394 humpback whales constitute the western North Pacific stock (Calambokidis *et al.* 1997). Waite *et al.* (1999) identified 127 individual humpback whales in the Kodiak Island region between 1991 and 1994 and estimated there were 651 whales in this region (95% CI: 356-1,523). Waite *et al.* (1999) also estimated that 200 humpback whales regularly feed in Prince William Sound. Subsequently, based on mark-recapture analysis of photo-identification studies, several investigators concluded that the central North Pacific stock consists of at least 4,000 humpback whales (Calambokidis *et al.* 1997, Ferrero *et al.* 2000). Other than these estimates of the size of the humpback whale population, the available information is not sufficient to determine population trends. In the BSAI, the humpback whale population was dramatically reduced by commercial whaling (see the discussion of commercial whaling in the Environmental Baseline chapter). The humpback whale population is believed to have increased since whaling ceased, although the rate of increase is unknown.

Estimates of the number of individuals in the Northern Pacific stock have recently risen. Estimates in the 1980's ranged from 1407 to 2,100 (Baker 1985; Darling and Morowitz 1986; Baker and Herman 1987), while recent estimates of abundances were approximately 6,000 in the North Pacific (Calambokidis *et al.* 1997; Cerchio 1998; Mobley *et al.* 1999b).

Studies based on resighting individuals through photographs estimate 6,010 animals (S.E. = 474) for the entire North Pacific (Calambokidis *et al.* 1997). The Central North Pacific stock of humpback whales winters in the waters of the main Hawaiian Islands and feeds on the summer grounds of Southeast Alaska and Prince William Sound. A population estimate of 1,407 whales was derived using capture-recapture methodology (95% CI: 1,113 - 1,701) for data collected in 1980-83 (Baker and Herman 1987).

Cerchio (1998) estimated that about 4,000 animals visit Hawaii annually. Aerial surveys conducted between 1976 and 1990 found a significant increase in sighting rates of humpbacks over that time (Mobley *et al.* 1999a), consistent with the increase in photographic estimates. Finally, aerial surveys using line-transect methodologies were conducted in 1993, 1995 and 1998. Hawaii population estimates derived from the sighting data show an increase from 2,717 animals (+/- 608) in 1993, to 3,284 (+/- 646) in 1995 and 3,852 (+/- 777) in 1998 (Mobley *et al.*

1999b).

New information has become available on the status and trends of the humpback whale population in the North Atlantic. Although current and maximum net productivity rates are unknown at this time, the population is apparently increasing. It has not yet been determined whether this increase is uniform across all six feeding stocks (Waring *et al. in prep.*). The rate of increase has been estimated at 9.0 percent (CV=0.25) by Katona and Beard (1990), while a 6.5 percent rate was reported for the Gulf of Maine by Barlow and Clapham (1997) using data through 1991. The rate reported by Barlow and Clapham (1997) may roughly approximate the rate of increase for the portion of the population within the action area. The best estimate of abundance for the North Atlantic humpback whale population is 10,600 animals (CV=0.067; Smith *et al.* 1999), while the minimum population estimate used for NMFS management purposes is 10,019 animals (CV = 0.067; Waring *et al. in prep.*). The Northeast Fisheries Science Center is considering recommending that NMFS identify the Gulf of Maine feeding stock as the management stock for this population in U.S. waters. A population estimate for the Gulf of Maine portion of the population is not available.

Impacts of human activity on this species

In the 1990s, no more than 3 humpback whales were killed annually in U.S. waters by commercial fishing operations in the Atlantic and Pacific Oceans. Between 1990 and 1997, no humpback whale deaths have been attributed to interactions with groundfish trawl, longline and pot fisheries in the Bering Sea, Aleutian Islands, and Gulf of Alaska (Hill and DeMaster 1999). Humpback whales have been injured or killed elsewhere along the mainland U.S. and Hawaii (Barlow *et al.* 1997). In 1991, a humpback whale was observed entangled in longline gear and released alive (Hill *et al.* 1997). In 1995, a humpback whale in Maui waters was found trailing numerous lines (not fishery-related) and entangled in mooring lines. The whale was successfully released, but subsequently stranded and was attacked and killed by tiger sharks in the surf zone. In 1996, a humpback whale calf was found stranded on Oahu with evidence of vessel collision (propeller cuts; NMFS unpublished data). Also in 1996, a vessel from Pacific Missile Range Facility in Hawaii rescued an entangled humpback, removing two crab pot floats from the whale; the gear was traced to a recreational fisherman in southeast Alaska. 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 in the North Pacific Ocean.

Humpback whales seem to respond to moving sound sources, such as whale-watching vessels, fishing vessels, recreational vessels, and low-flying aircraft (Beach and Weinrich 1989, Clapham *et al.* 1993, Atkins and Swartz 1989). Their responses to noise are variable and have been correlated with the size, composition, and behavior of the whales when the noises occurred (Herman *et al.* 1980, Watkins *et al.* 1981, Krieger and Wing 1986). Several investigators have suggested that noise may have caused humpback whales to avoid or leave feeding or nursery areas (Jurasz and Jurasz 1979b, Dean *et al.* 1985), while others have suggested that humpback whales may become habituated to vessel traffic and its associated noise. Still other researchers suggest that humpback whales may become more vulnerable to vessel strikes once they habituate to vessel traffic (Swingle *et al.* 1993; Wiley *et al.* 1995). In Hawaii, regulations prohibit boats from approaching within 91 m of adult whales and within 274 m in areas protected for mothers with a calf. Likewise, in Alaska, the number of cruise ships entering Glacier Bay has been limited to reduce possible disturbance.

Many humpback whales are killed by ship strikes along both coasts of the U.S. On the Pacific coast, a humpback whale is killed about every other year by ship strikes (Barlow *et al.* 1997). On the Atlantic coast, 6 out of 20 humpback whales stranded along the mid-Atlantic coast showed signs of major ship strike injuries (Wiley *et al.* 1995). Almost no information is available on the number of humpback whales killed or seriously injured by ship strikes outside of U.S. waters.

Right Whale

Species description and distribution

Historically, right whales occurred in all the world's oceans, from temperate to subarctic latitudes. The IWC currently recognizes two species of northern right whales: *Eubalaena glacialis* in the North Atlantic and *E. japonica* in the North Pacific. However, right whales in the North Atlantic, North Pacific, and the southern hemisphere of both oceans are currently listed under the ESA as one species: right whales (which includes E. *glaciali*s, *E. japonic*a, and E. *australi*s). For the purposes of ESA Section 7(a)(2) consultations, NMFS recognizes three major species of right whales: North Pacific, North Atlantic, and Southern Hemisphere that will be treated separately in the following discussion.

Pacific Right Whales

Very little is known of the size and distribution of right whales in the North Pacific and very few of these animals have been seen in the past 20 years. In 1996, a group of 3 to 4 right whales (which may have included a calf) were observed in the middle shelf of the Bering Sea, west of Bristol Bay and east of the Pribilof Islands (Goddard and Rugh 1998). In June 1998, a lone whale was observed on historic whaling grounds near Albatross Bank off Kodiak Island, Alaska (Waite and Hobbs 1999). Surveys conducted in July of 1997–2000 in Bristol Bay reported observations of lone animals or small groups of right whales in the same area as the 1996 sighting (Hill and DeMaster 1998, Perryman *et al.* 1999). Historical whaling records (Maury 1852, Townsend 1935, Scarff 1986) indicate the right whale ranged across the North Pacific above 35°N lat. They summered in the North Pacific Ocean and southern Bering Sea from April or May to September, with a peak in sightings in coastal waters of Alaska in June and July (Maury 1852, Townsend 1935, Omura 1958, Klumov 1962, Omura *et al.* 1969). Their summer range extended north of the Bering Strait (Omura *et al.* 1969). However, they were particularly abundant in the Gulf of Alaska from 145° to 151°W (Berzin and Rovnin 1966), and apparently concentrated in the Gulf of Alaska, especially south of Kodiak Islands and in the Eastern Aleutian Islands and southern Bering Sea shelf waters (Braham and Rice 1984).

The winter distribution patterns of right whales in the Pacific are virtually unknown, although some right whales have been sighted as far south as 27°N in the eastern North Pacific. They have also been sighted in Hawaii (Herman *et al.* 1980), California (Scarff 1986), Washington and British Columbia. Their migration patterns are unknown, but are believed to include north-south movements between summer and winter feeding areas. The scarcity of right whales is the result of an 800-year history of whaling that continued into the 1960s (Klumov 1962).

Atlantic Right Whales

NMFS recognizes two extant groups of right whales in the North Atlantic Ocean: an eastern population and a western population. A third subpopulation may have existed in the central Atlantic (migrating from east of Greenland to the Azores or Bermuda), but appears to be extinct (Perry *et al.,* 1999). In the 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). They are not found in the Caribbean and have been recorded only rarely in the Gulf of Mexico.

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 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.

Southern Right Whales

The IWC recognized six stock areas in the Southern Hemisphere (Figure 8)(Donovan 1991; Ohsumi and Kasamatsu 1986), which roughly correspond to the eight areas provisionally designated during the 1986 IWC Right Whale Workshop (IWC 1986a). The eight areas discussed by Brownell *et al.* (1986) are (1) Chilean coast (central and southern) (Anelio and Torres 1986); (2) Campbell and Auckland Islands; (3) New Zealand and the Kermadecs; (4) Southeast Australia (Tasmania, Victoria, eastern Australia, New South Wales, southern Australia); (5) Southwest Australia (from 135↗ E to 90↗ E); (6) Indian Ocean (sub-Antarctic island groups); (7) Eastern South Atlantic (South Africa to 20↗ E); (8) Western South Atlantic (20↗ E to South America).

Ohsumi and Kasamatsu (1986) reported high concentrations of right whales between the sub-tropical and Antarctic Convergences (Figure 9), with the highest density of sightings south of western Australia. This same Japanese sighting data indicated that the whales were found farthest south in January (the austral summer) and began moving north in February. This follows the seasonal residence patterns of whales studied in both South Africa and South America, where animals begin arriving on these wintering grounds from May through June, peaking in abundance during September, and then leaving these lower latitudes from December through January (Best and Scott 1993; Payne 1986).

Life history information

In both northern and southern hemispheres, right whales have been observed in the lower latitudes and more coastal waters during winter, and then tend to migrate to higher latitudes during the summer. Calving may occur in winter months when their distribution is more coastal, but the lack of sighting information suggests that calving may occur farther offshore. In summer and fall in both hemispheres, the distribution of right whales appears linked to the distribution of their principal zooplankton prey (Winn *et al.*1986). Essentially no information is available on the calving grounds or feeding habits of right whales in the North Pacific. Right whales in the North

Pacific are known to prey on a variety of zooplankton species including *Calanus plumchrus, C. cristatus, Euphausia pacifica, Metridia* spp., and copepods of the genus *Neocalanus.* This is similar to the feeding habits of right whales in the Gulf of Maine, which feed on zooplankton (primarily copepods) (see NMFS 1991b, Murison and Gaskin 1989). Right whales may compete with sympatric sei whales and many other predators or consumers of zooplankton in the eastern North Pacific and Bering Sea. Killer whales are suspected as possible predators, but no data from the North Pacific support this speculation (Scarff 1986).

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 (CeTAP 1982).

Northern right whales are mostly seen in groups of less than 12, most often singles or pairs (Jefferson *et al.* 1993). Larger groups may form on feeding or breeding grounds (Jefferson *et al.* 1993). In the North Pacific, most recent sightings have been of singles or pairs; however, two groups numbering six to ten and more than three whales were sighted in the northeastern Pacific (Goddard and Rugh 1998).

Vocalizations and hearing

A general description of the anatomy of the ear for cetaceans is provided in the description of the blue whale above. Limited data indicate that northern right whales produce moans of less than 400 Hz in frequency (Watkins and Schevill 1972; Thompson *et al.* 1979; Spero 1981). Right whales appear to use low frequency sounds as contact calls while summering in the Bay of Fundy (Spero 1981).

Listing status

Since 1949, the northern right whale has been protected from commercial whaling by the IWC. Right whales (both *E. glacialis* and *E. australi*s) are listed as endangered under the ESA. 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 June 3, 1994 (59 FR 28793). Critical habitat has not been designated for right whales in the North Pacific Ocean.

Population status and trends

The recovery plan for this species suggests that its population included more than 11,000 individuals before they were hunted, based on a known harvest of over 11,000 right whales by U.S. whalers with additional numbers struck and lost (Brownell *et al.* 1986). Current population estimates range from a low of 100−200 (Braham and Rice 1984) to a high of 220−500 (Berzin and Yablokov 1978 [in Berzin and Vladimirov 1981]), but Hill and DeMaster (1998) argue that it is not possible to reliably estimate the population size or trends of right whales in the North Pacific. As a result, no population projections are available for this species.

In the Atlantic Ocean, Knowlton *et al*. (1994) concluded, based on data from 1987 through 1992, that the western North Atlantic right whale population was growing at a net annual rate of 2.5% (CV = 0.12). This rate was also used in NMFS' marine mammal Stock Assessment Reports (e.g., Blaylock *et al*. 1995; Waring *et al.,* 1997). Since then, the data used in Knowlton *et al.* (1994) have been re-evaluated, and new attempts to model the trends of the western North Atlantic right whale population have been published (e.g., Kraus 1997; Caswell *et al*. 1999) and additional works are in progress (Caswell *et al.*, in prep; Wade and Clapham, in prep).

Caswell *et al.* (1999), using data on reproduction and survival through 1996, determined that the western North Atlantic right whale population was declining at a rate of 2.4% per year. One model they used suggested that the mortality rate of the right whale population has increased five-fold in less than one generation. According to Caswell *et al.* (1999), if the mortality rate as of 1996 does not decrease and the population performance does not improve, extinction could occur within 100 years and would be certain within 400 years, with a mean time to extinction of 191 years. In the three calving seasons following the Caswell *et al.* (1999) analysis, only 10 calves are known to have been born into the population. However, at least 16 calves (one of which subsequently died of unknown causes) were born during the 2000 to 2001 calving season, providing hope that the right whale's rate of decline may be slowing.

Impacts of human activity on this species

Before whaling began in the North Pacific Ocean, right whales were considered common or abundant in the North Pacific (Webb 1988). By 1900, observations of right whales in the North Pacific had become so rare it was impossible to know their population status or trend. In the Atlantic Ocean, the major known sources of anthropogenic mortality and injury of right whales include entanglement in commercial fishing gear and ship strikes.

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.

In the North Pacific, Scarff (1986) concluded that entanglement in fishing gear, noise, or continued hunting by countries who are not members of the IWC were not serious threats to right whales. However, Scarff (1986) argued that right whales in the North Pacific are particularly vulnerable to ship strikes and marine pollution because of their habit of feeding at, or near, the water surface.

Undersea exploration and development of mineral deposits, and the dredging of major shipping channels are continued threats to the coastal habitat of the right whale in both the North Atlantic and North Pacific. Offshore oil and gas activities have been proposed off the coast of the mid- and south- Atlantic U.S. and are currently being conducted in the Bering Sea and in eastern North Pacific.

In Russian waters, two fishery-related mortalities have been reported and offshore oil and gas development could potentially affect northern right whale habitat (Perry *et al.* 1999). Newly revealed Russian catch records show that approximately 3,212 southern right whales were harvested during the seasons 1948/49 through 1979/80.

These records are still incomplete, however, and no information on the geographic distribution of these catches has yet been revealed, although they occurred in both the North Pacific and the Southern Hemisphere (Zemsky *et al*. 1995).

Sei Whale

Species description and distribution

Sei whales are distributed in all of the world's oceans, except the Arctic Ocean. The IWC's Scientific Committee groups all of the sei whales in the entire North Pacific Ocean into one stock (Donovan 1991). However, some mark-recapture, catch distribution, and morphological research indicated that more than one stock exists; one between 175°W and 155°W longitude, and another east of 155° W longitude (Masaki 1976, 1977). During the winter, sei whales are found from 20°–23° N and during the summer from 35°–50° N (Masaki 1976, 1977). Horwood (1987) reported that 75–85% of the total North Pacific population of sei whales resides east of 180° longitude. In the North Pacific Ocean, sei whales have been reported primarily south of the Aleutian Islands, in Shelikof Strait and waters surrounding Kodiak Island, in the Gulf of Alaska, and inside waters of southeast Alaska (Nasu 1974, Leatherwood *et al.* 1982). Sei whales have been occasionally reported from the Bering Sea and in low numbers on the central Bering Sea shelf (Hill and DeMaster 1998). Masaki (1977) reported sei whales concentrating in the northern and western Bering Sea from July through September, although other researchers question these observations because no other surveys have ever reported sei whales in the northern and western Bering Sea. Horwood (1987) evaluated the Japanese sighting data and concluded that sei whales rarely occur in the Bering Sea.

The sei whale population in the western North Atlantic is assumed to consist of two stocks, a Nova Scotian Shelf stock and a Labrador Sea stock. Within the action area, the sei whale is most common on Georges Bank and into the Gulf of Maine/Bay of Fundy region during spring and summer, primarily in deeper waters. Individuals may range as far south as North Carolina. There are occasional influxes of this species 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. Although sei whales may prey upon small schooling fish and squid in the action area, available information suggests that calanoid copepods and euphausiids are the primary prey of this species.

Life history information

Reproductive activities for sei whales occur primarily in winter. Gestation is about 12.7 months and the calving interval is about 3 years (Rice 1977). Sei whales become sexually mature at about age 10 (Rice 1977). The age structure of the sei whale population is unknown. Rice (1977) estimated total annual mortality for adult females as 0.088 and adult males as 0.103. Andrews (1916) suggested that killer whales attacked sei whales less frequently than fin and blue whales in the same areas. Sei whales in the North Pacific feed on euphausiids and copepods, which make up about 95% of their diets (Calkins 1986). The balance of their diet consists of squid and schooling fish, including smelt, sand lance, Arctic cod, rockfish, pollack, capelin, and Atka mackerel (Nemoto and Kawamura 1977). Rice (1977) suggested that the diverse diet of sei whales may allow them greater opportunity to take advantage of variable prey resources, but may also increase their potential for competition with commercial fisheries. Endoparasitic helminths are commonly found in sei whales and can result in pathogenic effects when infestations occur in the liver and kidneys (Rice 1977).

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 m.

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

No studies have been published on the vocal behavior of sei whales. No studies have directly measured the sound sensitivity of sei whales (Croll *et al.* 1999). A general description of the anatomy of the ear for cetaceans is provided in the description of the blue whale above.

Listing status

In the North Pacific, the IWC began management of commercial taking of sei whales in 1970, and fin whales were given full protection in 1976 (Allen 1980). Sei whales were listed as endangered under the ESA in 1973. They are also protected by the Convention on International Trade in Endangered Species of wild flora and fauna and the MMPA. 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.

Population status and trends

Sei whale abundance prior to commercial whaling in the North Pacific has been estimated at 42,000 sei whales (Tillman 1977). Japanese and Soviet catches of sei whales in the North Pacific and Bering Sea increased from 260 whales in 1962 to over 4,500 in 1968 and 1969, after which the sei whale population declined rapidly (Mizroch *et al.* 1984). 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). Current abundance or trends are not known for stocks in the North Pacific. In California waters, only one confirmed and five possible sei whale sightings were recorded during 1991, 1992, and 1993 aerial and ship surveys (Carretta and

Forney 1993, Mangels and Gerrodette 1994). No sightings were confirmed off Washington and Oregon during recent aerial surveys. Several researchers have suggested that the recovery of right whales in the northern hemisphere has been slowed by other whales that compete with right whales for food. 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. 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 the larger sei whale population competes with the smaller right whale population and slows or prevents its recovery. The patterns in the eastern north Pacific Ocean: right whales and sei whales have overlapping foraging areas; right whales feed almost entirely on copepods, which sei whales prefer; and whalers depleted the population of right whales almost a century before they began to hunt sei whales (Rice 1974, Scarff 1986). Reeves *et al.* (1978) noted that several whale species feed on copepods in the eastern north Pacific, so the foraging patterns of sei whales may affect the foraging success of right whales.

There is insufficient data to determine trends of the sei whale population. Because there are no abundance estimates within the last 10 years, a minimum population estimate cannot be determined for NMFS management purposes (Waring *et al.* in prep.). Abundance surveys are problematic as this species is difficult to distinguish from the fin whale.

Impacts of human activity on this 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 eastern north Pacific, the sei whale population appeared to number about 40,000 animals until whaling began in 1963; by 1974, the sei whale population had been reduced to about 8,000 animals (Tilman 1977). No recent reports indicate sei whales are being killed or seriously injured as a result of fishing activities in any eastern North Pacific fishery (Perry *et al.* 1999). However, Barlow *et al.* (1997) note that a conflict may exist in the offshore drift gillnet fishery.

Sperm whale

Species description and distribution
Sperm whales are distributed in all of the world's oceans. Several authors have recommended three or more stocks of sperm whales in the North Pacific for management purposes (Kasuya 1991, Bannister and Mitchell 1980). However, the IWC's Scientific Committee designated two sperm whale stocks in the North Pacific: a western and eastern stock (Donovan 1991). The line separating these stocks 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: (1) Alaska, (2) California/Oregon/Washington, and (3) Hawaii. 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. Sperm whales are rarely found in waters less than 300 m in depth. They are often concentrated around oceanic islands in areas of upwelling, and along the outer continental shelf and mid-ocean waters. Because they inhabit deeper pelagic waters, their distribution does not include the broad continental shelf of the Eastern Bering Sea and these whales generally remain offshore in the eastern Aleutian Islands, Gulf of Alaska, and the Bering Sea.

A 1997 survey to investigate sperm whale stock structure and abundance in the eastern temperate North Pacific area did not detect a seasonal distribution pattern between the U. S. EEZ off California and areas farther west, out to Hawaii (Forney *et al*., 2000). A 1997 survey, which combined visual and acoustic line-transect methods, resulted in estimates of 24,000 (CV=0.46) sperm whales based on visual sightings, and 39,200 sperm whales (CV=0.60) based on acoustic detections and visual group size estimates (Forney *et al*., 2000). An analysis for the eastern tropical Pacific estimates abundance at 22,700 sperm whales (95% C. I. = 14,800-34,000; Forney *et al*., 2000).

For all stocks, the sperm whale is generally believed to engage in summer migrations, with mature males migrating north to the Gulf of Alaska, Aleutian Islands, and the Bering Sea, or south to the Antarctic. Females, calves and younger males, which usually remain below 40°N latitude in more tropical and temperate waters (Rice 1989), may be restricted in their migrations by an intolerance to low water temperatures. Mature males return to the warmer waters of the lower latitudes south of 40°N during the winter breeding season. Sperm whales may be found singly and in groups as large as fifty or more individuals, with solitary mature breeding males joining groups only during the breeding season (Gosho *et al.* 1984). During this time, sperm whales in the Pacific Ocean are usually distributed below 40°N Latitude. Historically, sperm whaling grounds in the Pacific were from 20°- 40°N and from 150°- 160°W and were located around the Hawaiian Islands, among other areas (Leatherwood *et al.* 1988).

Sperm whales have a strong preference for the 3,280 ft (1,000 m) depth contour and seaward. Berzin (1971) reported that they are restricted to waters deeper than 300 m (984 ft), while Watkins (1977) and Reeves and Whitehead (1997) reported that they are usually not found in waters less than 3,281 ft (1,000m) deep. While deep water is their typical habitat, sperm whales have been observed near Long Island, NY, in waters of 41-55 m (135-180 ft) (Scott and Sadove 1997). When 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 a good food supply (Clarke 1956). They can dive to depths of at least 2000 m (6562 ft), and may remain submerged for an hour or more (Watkins *et al.* 1993). Sperm whales feed primarily on buoyant, relatively slow-moving squid (Clark *et al.* 1993), but may also eat a variety of fish, including salmon (*Oncorhynchus* spp.), rockfish (*Sebastes* spp.), and lingcod (*Ophiodon elongatus*) (Caldwell and Caldwell 1983).

Sperm whales have been sighted in the Kauai Channel, the Alenuihaha Channel between Maui and the island of Hawaii, and off the island of Hawaii (Lee 1993; Mobley, *et al.*1999, Forney *et al*., 2000)). Additionally, the sounds of sperm whales have been recorded throughout the year off Oahu (Thompson and Friedl 1982). Twenty-one sperm whales were sighted during aerial surveys conducted in Hawaiian waters conducted from 1993 through 1998. Sperm whales sighted during the survey tended to be on the outer edge of a 50 - 70 km distance from the Hawaiian Islands, indicating that presence may increase with distance from shore (Mobley, pers. comm. 2000). However, from the results of these surveys, NMFS has calculated a minimum abundance of sperm whales within 46 km of Hawaii to be 43 individuals (Forney *et al.,* 2000). In the past five years, there is only one observed stranding of a sperm whale off Kauai which occurred in 1995 (NMFS, unpublished data).

In the Atlantic Ocean, NMFS' most recent stock assessment report notes that 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 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.

Life history information

Female sperm whales take about 9 years to become sexually mature (Kasuya 1991, as cited in Perry *et al.* 1999). Male sperm whales take between 9 and 20 years to become sexually mature, but will require another 10 years to become large enough to successfully compete for breeding rights (Kasuya 1991). Adult females give birth after about 15 months gestation and nurse their calves for 2 –3 years. The calving interval is estimated to be about four to six years (Kasuya 1991). The age distribution of the sperm whale population is unknown, but sperm whales are believed to live at least 60 years (Rice 1978). Estimated annual mortality rates of sperm whales are thought to vary by age, but previous estimates of mortality rate for juveniles and adults are now considered unreliable (IWC 1980, as cited in Perry *et al.* 1999). Sperm whales are known for their deep foraging dives (in excess of 3 km). They feed primarily on mesopelagic squid, but also consume octopus, other invertebrates, and fish (Tomilin 1967, Tarasevich1968, Berzin 1971). Perez (1990) estimated that their diet in the Bering Sea was 82% cephalopods (mostly squid) and 18% fish. Fish eaten in the North Pacific included salmon, lantern fishes, lancetfish, Pacific cod, pollack, saffron cod, rockfishes, sablefish, Atka mackerel, sculpins, lumpsuckers, lamprey, skates, and rattails (Tomilin 1967, Kawakami 1980, Rice 1986b). Sperm whales taken in the Gulf of Alaska in the 1960s had fed primarily on fish. Daily food consumption rates for sperm whales ranges from 2 - 4% of their total body weight (Lockyer 1976b, Kawakami 1980). Potential sources of natural mortality in sperm whales include killer whales and papilloma virus (Lambertson *et al.* 1987).

Diving and social behavior

Sperm whales are likely the deepest and longest diving mammal. Typical foraging dives last 40 min and descend to about 400m followed by approximately 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 is 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 1996b) 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 are on the order of 250 dB) with "shots" 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, changes in their abundance could affect the distribution and abundance of other marine species.

Listing status

Sperm whales have been protected from commercial harvest by the IWC since 1981, although the Japanese continued to harvest sperm whales in the North Pacific until 1988 (Reeves and Whitehead 1997). Sperm whales were listed as endangered under the ESA in 1973. 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.

Population status and trends

Current estimates for population abundance, status, and trends for the Alaska stock of sperm whales are not available (Hill and DeMaster 1999). Approximately 258,000 sperm whales in the North Pacific were harvested by commercial whalers between 1947 and 1987 (Hill and DeMaster 1999). In particular, the Bering Sea population of sperm whales (consisting mostly of males) was severely depleted (Perry *et al.* 1999). Catches in the North Pacific continued to climb until 1968, when 16,357 sperm whales were harvested. Catches declined after 1968 through limits imposed by the IWC.

The best abundance estimate that is currently available for the western North Atlantic sperm whale population is 2,698 (CV=0.67) animals, and the minimum population estimate used for NMFS management purposes is 1,617 (CV=0.67) (Waring *et al. in prep.*). Due to insufficient data, no information is available on population trends at this time for the western North Atlantic sperm whale stock. No information is available either on Mediterranean sperm whale population size or on the population relationship between sperm whales in the Mediterranean and the North Atlantic. However, the frequent observation of neonates in the Mediterranean and the scarcity of sightings from the Gibraltar area (Bayed and Beaubrun 1987) points to the possibility that sperm whales in the Mediterranean, like fin whales, may form a resident, reproductively isolated population.

Impacts of human activity on this species

In U.S. waters in the Pacific, sperm whales are known to have been incidentally taken only in drift gillnet operations, which killed or seriously injured an average of 9 sperm whales per year from 1991–95 (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 longlines 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. In 2000, the Japanese Whaling Association announced that it proposed to kill 10 sperm whales in the Pacific Ocean for research purposes, which was the first time sperm whales have been taken since the international ban on commercial whaling took effect in 1987. Despite protests from the U.S. government and members of the IWC, the Japanese government plans to conduct this “research.” The implications of this action for the status and trend of sperm whales is uncertain.

Green Sea Turtle

Species description and distribution

The genus *Chelonia* is composed of two taxonomic units at the population level, the eastern Pacific green turtle (referred to by some as “black turtle,” *C. mydas agassizii*), which ranges (including nesting) from Baja California south to Peru and west to the Galapagos Islands, and the nominate *C. m. mydas* in the rest of the range (insular tropical Pacific, including Hawaii).

Green turtles are distinguished from other sea turtles by their smooth carapace with four pairs of lateral scutes, a single pair of prefrontal scutes, and a lower jaw-edge that is coarsely serrated. Adult green turtles have a light to dark brown carapace, sometimes shaded with olive, and can exceed one meter in carapace length and 100 kilograms (kg) in body mass. Females nesting in Hawaii averaged 92 cm in straight carapace length (SCL), while at the Olimarao Atoll in Yap, females averaged 104 cm in curved carapace length (CCL) and approximately 140 kg. In the rookeries of Michoacán, Mexico, females averaged 82 cm in CCL, while males averaged 77 cm CCL (*in* NMFS and USFWS 1998a).

Green turtles are a circumglobal and highly migratory species that nest mainly in tropical and subtropical regions. The east Pacific green is also the second-most sighted turtle in the east Pacific during tuna fishing cruises; they are frequent along a north-south band from 15°N to 5°S along 90°W, and between the Galapagos Islands and Central American Coast (NMFS and USFWS 1998a). Their nonbreeding range is generally tropical, and can extend approximately 500-800 miles from shore in certain regions (Eckert 1993). They appear to prefer waters that usually remain around 20°C in the coldest month; for example, during warm spells (e.g., El Niño), green turtles may be found considerably north of their normal distribution. Stinson (1984) found green turtles to appear most frequently in U.S. coastal waters with temperatures exceeding 18°C.

Tag returns of eastern Pacific green turtles establish that these turtles travel long distances between foraging and nesting grounds. In fact, 75 percent of tag recoveries from 1982-90 were from turtles that had traveled more than 1,000 kilometers from Michoacán, Mexico. Even though these turtles were found in coastal waters, the species is not confined to these areas, as indicated by 1990 sightings records from a NOAA research ship. Observers have documented green turtles 1,000-2,000 statute miles from shore (Eckert 1993).

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). Hawaiian green turtles monitored through satellite transmitters were found to travel more than 1,100 km from their nesting beach in the French Frigate Shoals, south and southwest against prevailing currents to numerous distant foraging grounds within the 2,400 kilometer span of the archipelago (Balazs 1994; Balazs, *et al*. 1994; Balazs and Ellis 1996). Three green turtles outfitted with satellite tags on the Rose Atoll (the easternmost island at the Samoan Archipelago) traveled on a southwesterly course to Fiji, approximately 1,500 km distance (Balazs, *et al*. 1994).

In the western Atlantic, several major nesting assemblages have been identified and studied (Peters 1954, Carr and Ogren 1960, Parsons 1962, Pritchard 1969, Carr *et al.* 1978). In the continental United States, green turtle nesting occurs on the Atlantic coast of Florida (Ehrhart 1979). Occasional nesting has been documented along the Gulf coast of Florida, at Southwest Florida beaches, as well as the beaches on the Florida Panhandle (Meylan *et al.* 1995). Most documented green turtle nesting activity occurs on Florida index beaches, which were established to standardize data collection methods and effort on key nesting beaches. The pattern of green turtle nesting shows biennial peaks in abundance, with a generally positive trend during the ten years of regular monitoring since establishment of the index beaches in 1989, perhaps due to increased protective legislation throughout the Caribbean (Meylan *et al.* 1995).

Life history information

Based on growth rates observed in wild green turtles, skeletochronological studies, and capture-recapture studies, all in Hawaii, it is estimated that green turtles attain sexual maturity at an average age of at least 25 years (*in* Eckert 1993). Growth rates and age to first reproduction in other north Pacific populations remain unquantified (Eckert 1993). In Hawaii, green turtles lay up to six clutches of eggs per year (mean of 3.7), and clutches consist of about 100 eggs each. Females migrate to breed only once every two or possibly many more years. Eastern Pacific green turtles have reported nesting between two and six times during a season, laying a mean of between 65 and 86 eggs per clutch, depending on the area studied (Michoacan, Mexico and Playa Naranjo, Costa Rica (*in* Eckert 1993 and NMFS and USFWS 1998a).

Based on the behavior of post-hatchlings and juvenile green turtles raised in captivity, it is presumed 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 (NMFS and USFWS 1998a). The maximum recorded dive depth for an adult green turtle was 110 meters (Berkson 1967, *in* Lutcavage and Lutz 1997), while subadults routinely dive 20 meters for 9-23 minutes, with a maximum recorded dive of 66 minutes (Brill, *et al.* 1995, *in* Lutcavage and Lutz 1997). Additionally, it is presumed that drift lines or surface current convergences are preferential zones due to increased densities of likely food items. In the western Atlantic, 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. In the MHI these foraging and resting areas for adults usually occur at depths greater than 10 meters, but probably not normally exceeding 40 meters. Available information indicates that green turtle resting areas are in proximity to their feeding pastures (NMFS, 2000e). Immature Hawaiian green turtles have been found in increasing numbers residing in “foraging pastures” around the eight main Hawaiian Islands. These pastures consist of a narrow band of shallow water around these islands and “accounts for 96% of the benthic habitat potentially available for recruitment by post-pelagic green turtles” (Balazs 1996).

The majority portion of a green turtle’s life is spent on the foraging grounds. Green turtles are herbivores, and appear to prefer marine grasses and algae in shallow bays, lagoons and reefs (Rebel 1974). Some of the principal feeding pastures in the Gulf of Mexico include inshore south Texas waters, the upper west coast of Florida and the northwestern coast of the Yucatan

Peninsula. Additional important foraging areas in the western Atlantic include the Indian River Lagoon System 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). The preferred food in these areas are *Cymodocea, Thalassia, Zostera, Sagittaria,* and *Vallisneria* (Babcock 1937, Underwood 1951, Carr 1952, 1954).

Green sea turtles along the East Pacific coast seem to have a more carnivorous diet. Analysis of stomach contents of green turtles found off Peru revealed a large percentage of mollusks and polychaetes, while fish and fish eggs, and jellyfish and commensal amphipods comprised a lesser percentage (Bjorndal 1997). In the Hawaiian Islands, green turtles are site-specific and consistently feed in the same areas on preferred substrates, which vary by location and between islands (*in* Landsberg, *et al*. 1999).

Vocalizations and hearing

There is no information on green turtle vocalizations and hearing.

Listing status

Green sea 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. The IUCN has classified the green turtle as "endangered"

Population status and trends

The primary green turtle nesting grounds in the eastern Pacific are located in Michoacán, Mexico, and the Galapagos Islands, Ecuador (NMFS and USFWS 1998a). Here, green turtles were widespread and abundant prior to commercial exploitation and uncontrolled subsistence harvest of nesters and eggs. More than 165,000 turtles were harvested from 1965 to 1977 in the Mexican Pacific. In the early 1970s nearly 100,000 eggs per night were collected from these nesting beaches (*in* NMFS and USFWS 1998a). The nesting population at the two main nesting beaches in Michoacán (Colola, responsible for 70% of total green turtle nesting in Michoacán (Delgado and Alverado 1999) and Maruata) decreased from 5,585 females in 1982 to 940 in 1984. Despite long-term protection of females and their eggs at these sites since 1990, the population continues to decline, and it is believed that adverse impacts (including incidental take in various coastal fisheries as well as illegal directed take at forage areas) continue to prevent recovery of endangered populations (P. Dutton, NMFS, personal communication, 1999; W. Nichols, University of Arizona, personal communication, 2000). In addition, the black market for sea turtle eggs in Mexico has remained as brisk as before the ban (Delgado and Alvarado 1999). On Colola, an estimated 500-1,000 females nested nightly in the late 1960s. In the 1990s, that number dropped to 60-100 per night, or about 800-1,000 turtles per year (Eckert 1993). During the 1998-99 season, based on a comparison of nest counts and egg collection data, an estimated 600 greens nested at Colola. Although only about 5% of the nests were poached at Colola during this season, approximately 50% of the nests at Maruata were poached,

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[1] Under the IUCN, taxa are classified as endangered when they are not "critically endangered, but are facing a very high risk of extinction in the wild in the near future.

primarily because of difficulties in providing protections as a result of political infighting (Delgado and Alvarado 1999).

There are few historical records of abundance of green turtles from the Galapagos - only residents are allowed to harvest turtles for subsistence, and egg poaching occurs only occasionally. An annual average of 1,400 nesting females was estimated for the period 1976-1982 in the Galapagos Islands (NMFS and USFWS 1998a).

In the western Pacific, the only major (>2,000 nesting females) populations of green turtles occur in Australia and Malaysia. Pulau Redang, a coral fringed island located approximately 45 kilometers off the coast of Terengganu, Malaysia contains one of the largest green turtle rookeries in peninsular Malaysia, and a 1 nautical mile no-fishing zone has been established around the island to prevent interactions between fishing gear and internesting females (Liew and Chan 1994). Smaller colonies occur in the insular Pacific islands of Polynesia, Micronesia, and Malaysia (Wetherall *et al.* 1993). In Taiwan, Cheng and Chen (1996) report that between 1992 and 1994, green turtles were found nesting on 9 of 11 beaches on Wan-Am Island (Peng-Hu Archipelago). The numbers, however, were small, between 8 and 14 females nested during each of these 3 years. In Japan, the Ogasawara Islands, located approximately 1,000 km south of Tokyo, serve as the northern edge of green turtles rookeries. In the late 1800s, when Japan first colonized the islands, the government encouraged a sea turtle fishery. Declines in catch were steady from 1880-1890s (1,000-1,800 adults taken annually) through the mid-1920s (250 taken annually). Data from 1945-1972 (American occupation) indicate that 20-80 turtles were taken annually, and since then, annual harvests have fluctuated from 45-225 turtles per year (Horikoshi, *et al*. 1994) (Suganuma, *et al*. (1996) estimates 100 mating adults are speared by fishermen annually). Beach census data from 1985-93 indicate that 170-649 clutches were deposited each year (43 to 162 nesting females, assuming a female deposited 4 clutches during a nesting season). The Ogasawara population has declined in part due to past commercial exploitation, and it is likely to continue if fishery effort continues (Horikoshi, *et al*. 1994).

In Hawaii, green turtles nest on six small sand islands at French Frigate Shoals, a long atoll situated in the middle of the Hawaiian Archipelago (Balazs 1995). Unlike any other regional sea turtle populations, green turtles in Hawaii are genetically distinct and geographically isolated. Ninety percent of the nesting and breeding activity of the Hawaiian green turtle occurs at the French Frigate Shoals, where 200-700 females are estimated to nest annually (NMFS and USFWS 1998a). Important resident areas have been identified and are being monitored along the coastlines of Oahu, Molokai, Maui, Lanai, Hawaii, and at large nesting areas in the reefs surrounding the French Frigate Shoals, Lisianski Island, and Pearl and Hermes Reef (Balazs 1982; Balazs *et al.* 1987). Since the establishment of the ESA in 1973, and following years of exploitation, the nesting population of Hawaiian green turtles has shown a gradual but definite increase (Balazs 1996).

Impacts of human activities on this species

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). due to an "observed, estimated, inferred or suspected reduction of at least 50% over the last 10 years or three generations, whichever is longer," based on: (a) direct observation;

(b) an index of abundance appropriate for the species; and (c) actual or potential levels of exploitation.

Based on limited data, green turtle populations in the Pacific islands appear to have declined dramatically, because of harvests of green turtle eggs and adults by humans. In the green turtle recovery plans, directed take of eggs and turtles was identified as a “major problem” in the 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). Severe overharvests have resulted in modern times from a number of factors: 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).

The green turtle population in the northwest Hawaiian Islands is afflicted with a tumor disease, fibropapilloma, which is of an unknown etiology and often fatal, as well as spirochidiasis, both of which are the major causes of strandings of this species (G. Balazs, NMFS, personal communication, 2000). 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 that there is an association between the distribution of fibropapillomatosis in the Hawaiian 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 an inhibiting factor to the full recovery of the Hawaiian green turtle populations, and the incidence of decreased growth rates in afflicted turtles is a minimum estimate of the impact of the disease (Balazs, *et al*. 1998).

Hawksbill Sea Turtle

Species description and distribution

Hawksbill turtles are small turtles, with straight carapace lengths less than 95 cm (only Kemp’s ridley turtles are smaller). Hawksbill turtles occur in the Atlantic, Pacific, and Indian Oceans, the Red Sea, including waters around Australia, from California south to Peru, Japan, the British Isles, northwestern Africa, Madagascar, from Massachusetts to southern Brazil (Ernst and Barbour 1989).

Life history information

Hawksbill turtles nest on tropical islands and continental shorelines in tropical regions of the world. Hawksbill turtles mate in shallow water off their nesting beaches. Hawksbill turtles usually select nest sites under cover of woody vegetation, although they will build nests without such cover if it is not available. Hawksbill turtles tend to forage on sponges, although they have also been known to consume hydroids, coral, sea urchins, gastropods and bivalve mollusks, barnacles, crustaceans, and fish

Diving and social behavior

There are no known studies of deep diving in hawksbill turtles, based on their foraging behavior in the Caribbean region, hawksbill turtles probably forage at depths of 100 meters or greater (the dominant sponge communities in the Caribbean are found at depths of 38 to 76 meters).

Vocalizations and hearing

There is no information on hawksbill turtle vocalizations and hearing

Listing status

Hawksbill turtles were listed as endangered on June 2 1970, under the Endangered Species Conservation Act of 1969. The coastal waters surrounding Mona and Monito Islands, off the west coast of Puerto Rico, have been designated critical habitat for hawksbill turtles.

Population status and trends

Of the 65 geopolitical units worldwide, where estimates of relative hawksbill nesting density exists, 38 of them have hawksbill populations that are suspected or known to be in decline and an additional 18 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).

Impacts of human activities on this species

The recovery plan for hawksbill turtles in the Pacific Ocean identified twenty-seven threats to the species, that include directed take of eggs, capture and killing of adults, human encroachment on coastal ecosystems, predation of nests, beach erosion, artificial lighting, and degradation of their marine habitats.

Kemp's Ridley Sea Turtle

Species description and distribution

The Kemp's ridley is the most endangered of the world's sea turtle species. The are distributed in the western Atlantic Ocean from Nova Scotia and, possibly, Newfoundland, south to Bermuda, west through the Gulf of Mexico to Mexico. They have also been recorded to cross the Gulf Stream to England, Ireland, the Scilly Islands, France, the Azores, and the Mediterranean Sea. Almost all nesting occurs in the southern coast of Tamaulipas, Mexico, near Rancho Nuevo.

Life history information

Juvenile Kemp's ridleys use northeastern and mid-Atlantic coastal waters of the U.S. Atlantic coastline as primary developmental habitat during summer months, with shallow coastal embayments serving as important foraging grounds. Post-pelagic ridleys feed primarily on crabs, consuming a variety of species, including *Callinectes* sp., *Ovalipes* sp., *Libinia* sp., and *Cancer* sp. Mollusks, shrimp, and fish are consumed less frequently (Bjorndal 1997). Juvenile ridleys migrate south as water temperatures cool in fall, and are predominantly found in shallow coastal embayments along the Gulf Coast during fall and winter months.

Kemp's ridley turtles found in mid-Atlantic waters are primarily post-pelagic juveniles averaging 40 centimeters in carapace length, and weighing less than 20 kilograms (Terwilliger and Musick 1995). Next to loggerheads, they are the second most abundant sea turtle in Virginia and Maryland waters, arriving in these areas during May and June, and migrating to more southerly waters from September to November (Keinath *et al.* 1987; Musick and Limpus 1997). In the Chesapeake Bay, ridleys frequently forage in shallow embayments, particularly in areas supporting submerged aquatic vegetation (Lutcavage and Musick 1985; Bellmund *et al.* 1987; Keinath *et al.* 1987; Musick and Limpus 1997). The juvenile population in Chesapeake Bay is estimated to be 211 to 1,083 turtles (Musick and Limpus 1997).

Juvenile ridleys follow regular coastal routes during spring and fall migrations to and from developmental foraging grounds along the mid-Atlantic and northeastern coastlines. Consequently, many ridleys occurring in coastal waters off Virginia and Maryland are transients involved in seasonal migrations. However, Maryland's and Virginia's coastal embayments - which contain an abundance of crabs, shrimp, and other prey as well as preferred foraging habitat such as shallow subtidal flats and submerged aquatic vegetation beds - are likely used as a foraging ground by Kemp's ridley sea turtles (John Musick, Virginia Institute of Marine Science, 1998 personal communication; Sherry Epperly, National Marine Fisheries Service, Beaufort Laboratory, Beaufort North Carolina, 1998 personal communication; Molly Lutcavage, New England Aquarium 1998 personal communication). No known nesting occurs on Virginia or Maryland beaches.

Vocalizations and hearing

There is no information on Kemp's ridley turtle vocalizations and hearing

Listing status

Kemp's ridley sea turtles are listed as endangered under the ESA. The International Union for Conservation of Nature and Natural Resources has classified the Kemp's ridley turtle as "endangered."

Population status and trends

The only major nesting site for ridleys is a single stretch of beach near Rancho Nuevo, Tamaulipas, Mexico (Carr 1963). Estimates on the adult population reached a low of 1,050 in 1985, and increased to 3,000 individuals in 1997. First-time nesting adults increased from 6 percent to 28 percent from 1981 to 1989, and from 23 percent to 41 percent from 1990 to 1994, indicating that the ridley population may be in the early stages of exponential growth (TEWG 1998).

Impacts of human activities on this species

Anthropogenic impacts to the Kemp's ridley population are similar to those discussed above for the loggerhead sea turtle. Mortality in the large juvenile and adult life stage would have the greatest impact to the Kemp's ridley population (TEWG 1998). The vast majority of ridleys identified along the Atlantic Coast have been juveniles and subadults. Loss of individuals, particularly large juveniles, in the Atlantic resulting from human activities may therefore impede recovery of the Kemp's ridley sea turtle population. Sea sampling coverage in the northeast otter trawl fishery, pelagic longline fishery, and southeast shrimp and summer flounder bottom trawl fisheries has recorded takes of Kemp's ridley turtles. As with loggerheads, a large number of

Kemp's ridley turtles are captured and killed in the southeast shrimp fishery each year and may be captured and killed in northeast shrimp fishery and bottom longline fisheries.

Leatherback Sea Turtle

Species Description and Distribution

The leatherback is the largest living turtle. Leatherback sea turtles are widely distributed throughout the oceans of the world, and are found throughout waters of the Atlantic, Pacific, Caribbean, and the Gulf of Mexico (Ernst and Barbour 1972). In the Pacific Ocean, they range as far north as Alaska and the Bering Sea and as far south as Chile and New Zealand. In Alaska, leatherback turtles are found as far north as 60.34 N, 145.38W and as far west as the Aleutian Islands (Hodge 1979, Stinson 1984). Leatherback turtles have been found in the Bering Sea along the coast of Russia (Bannikov *et al*. 1971).

Leatherback turtles undertake the longest migrations of any other sea turtle and exhibit the broadest thermal tolerances (NMFS and USFWS 1998). Leatherback turtles are able to inhabit intensely cold waters for a prolonged period of time because leatherbacks are able to maintain body temperatures several degrees above ambient temperatures. Leatherback turtles are typically associated with continental shelf habitats and pelagic environments, and are sighted regularly in offshore waters (>328 ft). Leatherback turtles regularly occur in deep waters (>328 ft), and an aerial survey study in the north Atlantic Ocean sighted leatherback turtles in water depths ranging from 3 to 13,618 ft, with a median sighting depth of 131.6 ft (CeTAP 1982). This same study found leatherbacks in waters ranging from 7 to 27.2°C.

Leatherback turtles are uncommon in the insular Pacific Ocean, but individual leatherback turtles are sometimes encountered in deep water and prominent archipelagoes. To a large extent, the oceanic distribution of leatherback turtles may reflect the distribution and abundance of their macroplanktonic prey, which includes medusae, siphonophores, and salpae in temperate and boreal latitudes (NMFS and USFWS 1996). There is little information available on their diet in subarctic waters.

Life History Information

Although leatherbacks are a long lived species (> 30 years), they are somewhat faster to mature than loggerheads, with an estimated age at sexual maturity reported as about13-14 years for females, and an estimated minimum age at sexual maturity of 5-6 years, with 9 years reported as a likely minimum (Zug and Parham 1996).

Leatherback sea turtles are predominantly distributed pelagically where they feed on jellyfish such as *Stomolophus, Chryaora,* and *Aurelia* (Rebel 1974). Leatherbacks are deep divers, with recorded dives to depths in excess of 1000 m, but they may come into shallow waters if there is an abundance of jellyfish nearshore. They also occur annually in places such as Cape Cod and Narragansett bays during certain times of the year, particularly the fall.

Some of the largest nesting populations of leatherback turtles in the world border the Pacific Ocean, but no nesting occurs on beaches under U.S. jurisdiction. However, the Pacific coast of Mexico is generally regarded as the most important breeding ground for nesting leatherback turtles in the world. Leatherback turtles do not generally nest in the insular Central and North Pacific (except the Solomon Islands, Vanuatu, and Fiji). Nesting is widely reported from the western Pacific, including China, southeast Asia, Indonesia, and Australia.

Listing status

The leatherback was listed as endangered on June 2, 1970 and a recovery plan was issued in 1998. Leatherback turtles are included in Appendix I of the Convention on International Trade in Endangered Species of Wild Fauna and Flora, which effectively bans trade. Critical habitat has not been designated for leatherback turtles in the U.S. Pacific, largely because nesting is not known to occur in U.S. territory and important foraging areas have not been identified.

Population status and trends

Globally, leatherback turtle populations have been decimated worldwide. The global leatherback turtle population was estimated to number approximately 115,000 adult females in 1980 (Pritchard 1982), but only 34,500 in 1995 (Spotila *et al*. 1996). The decline can be attributed to many factors including fisheries as well as intense exploitation of the eggs (Ross 1979). On some beaches nearly 100% of the eggs laid have been harvested (Eckert 1996). Eckert (1996) and Spotila *et al*. (1996) record that adult mortality has also increased significantly, particularly as a result of driftnet and longline fisheries.

The Pacific population appears to be in a critical state of decline. The East Pacific leatherback population was estimated to be over 91,000 adults in 1980 (Spotila 1996), but is now estimated to number less than 3,000 total adult and subadult animals (Spotila 2000). Leatherback turtles have experienced major declines at all major Pacific basin rookeries. At Mexiquillo, Michoacan, Mexico, Sarti *et al*. (1996) reported an average annual decline in nesting of about 23% between 1984 and 1996. The total number of females nesting on the Pacific coast of Mexico during the 1995-1996 season was estimated at fewer than 1,000. Less than 700 females are estimated for Central America (Spotila 2000). In the western Pacific, the decline is equally severe. Current nestings at Terengganu, Malaysia represent 1% of the levels recorded in the 1950s (Chan and Liew 1996).

The status of the Atlantic population is less clear. In 1996, it was reported to be stable, at best (Spotila 1996), but numbers in the Western Atlantic at that writing were reported to be on the order of 18,800 nesting females. According to Spotila (pers. com.), the Western Atlantic population currently numbers about 15,000 nesting females, whereas current estimates for the Caribbean (4,000) and the Eastern Atlantic (i.e. off Africa, numbering ~ 4,700) have remained consistent with numbers reported by Spotila *et al*. in 1996. Between 1989 and 1995, marked leatherback returns to the nesting beach at St. Croix averaged only 48.5%, but that the overall nesting population grew (McDonald, et. al 1993). This is in contrast to a Pacific nesting beach at Playa Grande, Costa Rica, where only 11.9% of turtles tagged in 1993-94 and 19.0% of turtles tagged in 1994-95 returned to nest over the next five years. Characterizations of this population suggest that it has a very low likelihood of survival and recovery in the wild under current conditions.

Spotila *et al*. (1996) describe a hypothetical life table model based on estimated ages of sexual maturity at both ends of the species' natural range (5 and 15 years). The model concluded that leatherbacks maturing in 5 years would exhibit much greater population fluctuations in response to external factors than would turtles that mature in 15 years. Furthermore, the simulations indicated that leatherbacks could maintain a stable population only if both juvenile and adult survivorship remained high, and that if other life history stages (i.e. egg, hatchling, and juvenile) remained static, “stable leatherback populations could not withstand an increase in adult mortality above natural background levels without decreasing.

Impacts of human activity on this species

The primary threats to leatherback turtles are entanglement in fishing gear (e.g., gillnets, longlines, lobster pots, weirs), boat collisions, and ingestion of marine debris (NMFS and USFWS 1997). The foremost threat is the number of leatherback turtles killed or injured in fisheries. Spotila (2000) states that a conservative estimate of annual leatherback fishery-related mortality (from longlines, trawls and gillnets) in the Pacific 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. As noted above, leatherbacks normally live at least 30 years, usually maturing at about 12-13 years. Such long-lived species can not withstand such high rates of anthropogenic mortality.

Based on recent modeling efforts, the leatherback turtle population cannot withstand more than a 1% human-related mortality level which translates to 150 nesting females (Spotila *et al*. 1996; Spotila pers. comm.). As noted previously, there are many human-related sources of mortality to leatherbacks; every year, 1,800 leatherback turtles are expected to be captured or killed as a result of federally-managed activities in the U.S. (this total includes both lethal and non-lethal take). An unknown number of leatherbacks are captured or killed in fisheries managed by states. Spotila *et al*. (1996) recommended not only reducing fishery-related mortalities, but also advocated protecting eggs and hatchlings. Zug and Parham (1996) point out that a combination of the loss of long-lived adults in fishery-related mortalities and a lack of recruitment stemming from elimination of annual influxes of hatchlings because of intense egg harvesting has caused the sharp decline in leatherback populations.

Loggerhead Sea Turtle

Species description and distribution

Loggerhead sea turtles have reddish brown, bony carapaces, with large heads (up to 25 cm wide in some adults). Adults loggerhead turtles typically weigh between 80 and 150 kg, with average lengths between 95-100 cm (curved carapace length or CCL; *in* Dodd 1988, Limpus 1985).

Loggerhead turtles are a cosmopolitan species, found in temperate and subtropical waters and inhabiting pelagic waters, continental shelves, bays, estuaries and lagoons. The species is divided into five populations: the Atlantic Ocean, Pacific Ocean, Indian Ocean, Caribbean Sea and Mediterranean Sea populations. These populations are further divided into nesting aggregations. In the Pacific Ocean, loggerhead turtles are represented by a northwestern Pacific nesting aggregation (located in Japan) and a smaller southwestern nesting aggregation that occurs in Australia (Great Barrier Reef and Queensland), New Caledonia, New Zealand, Indonesia, and Papua New Guinea. In the western Atlantic Ocean, NMFS recognizes five major nesting aggregations: (1) a northern nesting aggregation that occurs from North Carolina to northeast Florida, about 29° N; (2) a south Florida nesting aggregation, occurring from 29° N on the east coast to Sarasota on the west coast; (3) a Florida panhandle nesting aggregation, occurring at Eglin Air Force Base and the beaches near Panama City, Florida;(4) a Yucatán nesting aggregation, occurring on the eastern Yucatán Peninsula, Mexico and (5). In addition, Atlantic and Caribbean nesting aggregations are found in Honduras, Colombia, Panama, the Bahamas, and Cuba. In the Mediterranean Sea, nesting aggregations in Greece, Turkey, Israel, Italy, and several other sites have been recorded. One of the largest loggerhead nesting aggregations in the world is found in Oman on the Indian Ocean.

Life history information

The life cycle of loggerhead can be divided into seven stages (this characterization follows the work of Crouse *et al.* 1987; other authors have proposed other divisions): eggs and hatchlings; small juveniles, large juveniles, subadults, novice breeders, first year remigrants, and mature breeders, each with their own distribution, duration, and vital rates.

Hatchling loggerhead sea turtles migrate out to the ocean where they are generally believed to lead a pelagic existence for as long as 7-12 years. When they complete their pelagic stage (generally, when they are 10 to 25 years of age) loggerhead turtles shift to a benthic habit. Benthic immature loggerhead turtles forage in the open ocean and coastal areas along continental shelves, bays, lagoons, and estuaries (NMFS and USFWS 1991b). When they are 21 to 35 years of age, loggerhead turtles become sexually mature, although the age at which they reach maturity will vary widely among populations (Frazer and Ehrhart 1985; Frazer and Limpus 1998).

Adult loggerhead turtles are known to make considerable migrations between foraging areas and nesting beaches (TEWG 1998). Of all sea turtle species, loggerheads are the most temperate and subtropical in their nesting habits. Adult, female loggerheads are iteroparous both within and among years, typically displaying high fecundity (NMFS and USFWS 1995).

Vocalizations and hearing

There is no information on loggerhead turtle vocalizations and hearing

Listing status

Loggerhead turtles were listed as threatened under the Endangered Species Act of 1973 on July 28, 1978, but they are considered endangered by the World Conservation Union and under the Convention of International Trade in Endangered Species of Flora and Fauna.

Population status and trends

The NMFS SEFSC report (2001) summarizes trend analyses for number of nests sampled from beaches for the northern subpopulation and the South Florida subpopulation and concluded that from 1978-1990, the northern subpopulation has been stable at best and possibly declining (less than 5% per year). From 1990 to the present, the number of nests in the northern subpopulation has been increasing at 2.8%-2.9% annually; however, there are confidence intervals about these estimates that include no growth (0%). Over the same time frame, the South Florida population has been increasing at 5.3%-5.4% per year from 1978-1990, and increasing at 3.9%-4.2% since 1990.

From a global perspective, the southeastern U.S. nesting aggregation is a critical component of this species. It is second in size only to the nesting aggregations in the Oman and represents about 35% and 40% of the nesting of this species globally. The status of the Oman nesting beaches has not been evaluated recently, but they are located in a part of the world that has a history of periodic, disruptive, events (*e.g.,* political upheavals, wars, and catastrophic oil spills). The resulting risk facing this nesting aggregation and these nesting beaches is cause for considerable concern (Meylan *et al*., 1995).

Impacts of human activities on this species

Loggerhead turtles are affected by the same human activities that were reported for the other sea turtles discussed previously. The major factors inhibiting their recovery include mortalities caused by fishery interactions and degradation of the beaches on which they nest. Loggerhead sea turtles face a number of threats in the marine environment, including oil and gas exploration, development, and transportation; marine pollution; trawl, purse seine, hook and line, gill net, pound net, longline, and trap fisheries; underwater explosions; dredging, offshore artificial lighting; power plant entrapment; entanglement in debris; ingestion of marine debris; marina and dock construction and operation; boat collisions; and poaching.

Several published reports have presented the problems facing long-lived species that delay sexual maturity in a world replete with threats from a modern, human population (Congdon *et al.* 1993, Congdon and Dunham 1994, Crouse *et al.* 1987, Crowder *et al.* 1994, Crouse 1999). In general, these reports concluded that animals that delay sexual maturity and reproduction must have high, annual survival as juveniles through adults to ensure that enough juvenile sea turtles survive to reproductive maturity and then reproduce enough times to maintain stable population sizes. This general rule applies to sea turtles, particularly loggerhead sea turtles, because the rule originated in studies of sea turtles (Crouse *et al.* 1987, Crowder *et al.* 1994, Crouse 1999). Heppell *et al.* (in prep.) specifically showed that the growth of the loggerhead sea turtle population was particularly sensitive to changes in the annual survival of both juvenile and adult sea turtles and that the adverse effects of the pelagic longline fishery on loggerheads from the pelagic immature phase appeared critical to the survival and recovery of the species. Crouse (1999) concluded that relatively small changes in annual survival rates of both juvenile and adult loggerhead sea turtles will adversely affect large segments of the total loggerhead sea turtle population.

Loggerhead sea turtles also face numerous threats from natural causes. For example, there is a significant overlap between hurricane seasons in the Caribbean Sea and northwest Atlantic Ocean (June to November) and loggerhead sea turtle nesting season (March to November); hurricanes can have potentially disastrous effects on the survival of eggs in sea turtle nests. For example, in 1992, Hurricane Andrew affected turtle nests over a 90-mile length of coastal Florida; all of the eggs were destroyed by storm surges on beaches that were closest to the eye of this hurricane (Milton *et al*. 1992).

Olive Ridley Sea Turtle

Species description and distribution

Olive ridley turtles are the smallest sea turtle and occur in the tropical waters of the Pacific and Indian Oceans from Micronesia, Japan, India, and Arabia south to northern Australia and southern Africa. In the Atlantic Ocean off the western coast of Africa and the coasts of northern Brazil, French Guiana, Surinam, Guyana, and Venezuela in South America, and occasionally in the Caribbean Sea as far north as Puerto Rico. In the eastern Pacific Ocean, olive ridley turtles are found from the Galapagos Islands north to California. While olive ridley turtles have a generally tropical to subtropical range, individual turtles have been reported as far as the Gulf of Alaska (Hodge and Wing, 2000).

Olive ridley turtles nest along continental margins and oceanic islands. The largest nesting aggregation in the world occurs in the Indian Ocean along the northeast coast of India where more than 600,000 olive ridley turtles nested in a single week in 1991 (Mrosovsky 1993). The second most important nesting area occurs in the eastern Pacific along the west coast of Mexico and Central America. Olive ridley turtles also nest along the Atlantic coast of South America, western Africa, and the western Pacific (Sternberg 1981, Groombridge 1982).

In the eastern Pacific, olive ridley turtles nest along the Mexico and Central American coast, with large nesting aggregations occurring at a few select beaches located in Mexico and Costa Rica. Few turtles nest as far north as southern Baja California, Mexico (Fritts *et al*. 1982) or as far south as Peru (Brown and Brown 1982). The post-nesting migration routes of olive ridleys traversed thousands of kilometers of deep oceanic waters, ranging from Mexico to Peru, and more than 3,000 kilometers out into the central Pacific (Plotkin, *et al*. 1993). Although they are the most abundant north Pacific sea turtle, surprisingly little is known of the oceanic distribution and critical foraging areas of olive ridley turtles.

Most records of olive ridley turtles are from protected, relative shallow marine waters. Deraniyagalia (1939) described the habitat of olive ridley turtles as shallow waters between reefs and shore, larger bays, and lagoons. Nevertheless, olive ridley turtles have also been observed in the open ocean. Since, olive ridley turtles throughout the eastern Pacific Ocean depend on rich upwelling areas off South America for food, olive ridley turtles sighted offshore may have been foraging.

Life history information

Olive ridley turtles begin to aggregate near the nesting beach two months before the nesting season, and most mating is generally assumed to occur in the vicinity of the nesting beaches, although copulating pairs have been reported over 100 km from the nearest nesting beach. Olive ridleys reach sexual maturity between 8 and 10 years of age, and approximately 3 percent of the number of hatchlings recruit to the reproductive population (Marquez 1982 and Marquez 1992, in Salazar, *et al*. 1998). Adult, female olive ridley turtles reproduce until they are 21 years of age (Pandav and Kar 2000).

The mean clutch size for females nesting on Mexican beaches is 105 eggs, in Costa Rica, clutch size averages between 100 and 107 eggs (in NMFS and USFWS 1998d). Females generally lay 1.6 clutches of eggs per season by Mexico (Salazar, *et al.* 1998) and two clutches of eggs per season in Costa Rica (Eckert 1993). Data on the remigration intervals of olive ridleys in the eastern Pacific are scarce; however, in the western Pacific (Orissa, India), females showed an annual mean remigration interval of 1.1 years.

Like leatherback turtles, most olive ridley turtles lead a primarily pelagic existence (Plotkin *et al*. 1993), migrating throughout the Pacific, from their nesting grounds in Mexico and Central America to the north Pacific. Olive ridley turtles forage throughout the eastern tropical Pacific Ocean, often in large groups, or flotillas, and are occasionally found entangled in scraps of net or other floating debris. In a three year study of communities associated with floating objects in the eastern tropical Pacific, Arenas and Hall (1992) found sea turtles in 15 percent of observations and suggested that flotsam may provide the turtles with food, shelter, and/or orientation cues in otherwise featureless marine landscapes. Olive ridleys feed on tunicates, salps, crustaceans, other invertebrates and small fish.

Diving and social behavior

Although olive ridley turtles are probably surface feeders, they have been caught in trawls at depths of 80-110 meters (NMFS and USFWS 1998), and a post-nesting female reportedly dove to a maximum depth of 290 meters. The average dive length for an adult female and adult male is reported to be 54.3 and 28.5 minutes, respectively (Plotkin 1994, in Lutcavage and Lutz 1997).

Vocalizations and hearing

There is no information on Olive ridley turtle vocalizations and hearing

Listing status

Olive ridley turtle populations on the Pacific coast of Mexico are listed as endangered under the ESA; all other populations are listed as threatened. The International Union for Conservation of Nature and Natural Resources has classified the Kemp's ridley turtle as "endangered" (IUCN Red List, 2000).

Population status and trends

Where population densities are high enough, nesting takes place in synchronized aggregations known as arribadas. The largest known arribadas in the eastern Pacific are off the coast of Costa Rica (~475,000 - 650,000 females estimated nesting annually) and in southern Mexico (~800,000 nests per year at La Escobilla, in Oaxaca, Mexico (Millá án, 2000)). In Costa Rica, 25,000 to 50,000 olive ridleys nest at Playa Nancite and 450,000 to 600,000 turtles nest at Playa Ostional each year (NMFS and USFWS 1998d). In an 11-year review of the nesting at Playa Ostional, (Ballestero, *et al*., 2000) report that the data on numbers of nests deposited is too limited for a statistically valid determination of a trend; although the number of nesting turtles has appeared to decline over a six-year period.

At a nesting site in Costa Rica, an estimated 0.2 percent of 11.5 million eggs laid during a single arribada produced hatchlings (in NMFS and USFWS 1998d). In addition, some female olive ridleys nesting in Costa Rica have been found afflicted with the fibropapilloma disease (Aguirre, *et al*. 1999). At Playa La Flor, the second most important nesting beach for olive ridleys on Nicaragua, Ruiz (1994) documented 6 arribadas (defined as 50 or more females resting simultaneously). The main egg predators were domestic dogs and vultures (*Coragyps atratus* and *Cathartes aura*).

In the western Pacific, information on the size of olive ridley nesting aggregations are limited although they do not appear to be recovering (with the exception of the nesting aggregation at Orissa, India). There are a few sightings of olive ridleys from Japan, but no report of egg-laying. Similarly, there are no nesting records from China, Korea, the Philippines, Taiwan, Viet Nam, or Kampuchea and nesting records in Indonesia are not sufficient to assess population trends (Eckert 1993, Suwelo 1999). In Thailand, olive ridleys occur along the southwest coast, on the Surin and Similan islands, and in the Andaman Sea. On Phra Thong Island, on the west coast of Thailand, the number of nesting turtles have declined markedly from 1979 to 1990.

Olive ridley turtles have been observed in Indonesia and surrounding waters, and some olive ridley turtles have been documented as nesting in this region recently. On Jamursba-Medi beach, on the northern coast of Irian Jaya, 77 olive ridley nests were documented from May to October, 1999 (Teguh 2000 in Putrawidjaja 2000).

Olive ridleys nest on the eastern and western coasts of peninsular Malaysia; however, nesting has declined rapidly in the past decade. The highest density of nesting was reported to be in Terengganu, Malaysia, and at one time yielded 240,000 eggs (2,400 nests, with approximately 100 eggs per nest; see Siow and Moll 1982, in Eckert 1993), while only 187 nests were reported from the area in 1990 (Eckert 1993). In eastern Malaysia, olive ridleys nest very rarely in Sabah and only a few records are available from Sarak (in Eckert 1993).

Olive ridleys are the most common species found along the east coast of India, migrating every winter to nest en-masse at three major rookeries in the state of Orissa, Gahirmatha, Robert Island, and Rushikulya (Pandav and Choudhury 1999). According to Pandav and Choudhury (1999), the number of nesting females at Gahirmatha has declined in recent years, although after three years of low nestings, the 1998-1999 season showed an increasing trend (Noronha *Environmental News Service*, April 14, 1999), and the 1999-2000 season had the largest recorded number of olive ridleys nesting in 15 years (*The Hindu*, March 27, 2000; *The Times of India*, November 15, 2000). During the 1996-1997 and 1997-98 seasons, there were no mass nestings of olive ridleys. During the 1998-1999 nesting season, around 230,000 females nested during the first arribada, lasting approximately a week (Pandav and Kar 2000); unfortunately, 80% of the eggs were lost due to inundation and erosion (B. Pandav, personal communication, in Shanker and Mohanty 1999). During 1999-2000, over 700,000 olive ridleys nested at Nasi islands and Babubali island, in the Gahirmatha coast.

Impacts of human activities on this species

In India, uncontrolled mechanized fishing in areas of high sea turtle concentration, primarily illegally operated trawl fisheries, has resulted in large scale mortality of adult olive ridley turtles during the last two decades. Since 1993, more than 50,000 olive ridleys have stranded along the coast, at least partially because of near-shore shrimp fishing (Shanker and Mohanty 1999). Fishing in coastal waters off Gahirmatha was restricted in 1993 and completely banned in 1997 with the formation of a marine sanctuary around the rookery. However, mortality due to shrimp trawling reached a record high of 13,575 ridleys during the 1997-1998 season and none of the approximately 3,000 trawlers operating off the Orissa coast use turtle excluder devices in their nets despite mandatory requirements passed in 1997 (Pandav and Choudhury 1999).

Historically, an estimated 10 million olive ridleys inhabited the waters in the eastern Pacific off Mexico (Cliffton *et al*. 1982 in NMFS and USFWS 1998). However, human-induced mortality caused this population to decline. From the 1960s to the 1970s, several million adult olive ridleys were harvested by Mexico for commercial trade with Europe and Japan. (NMFS and USFWS 1998). Although olive ridley meat is palatable, it was not widely sought after; its eggs, however, are considered a delicacy. Fisheries for olive ridley turtles were also established in Ecuador during the 1960s and 1970s to supply Europe with leather. (Green and Ortiz-Crespo 1982).

The nationwide ban on commercial harvest of sea turtles in Mexico, enacted in 1990, has improved the situation for the olive ridley. Surveys of important olive ridley nesting beaches in Mexico indicate increasing numbers of nesting females in recent years (Marquez *et al*. 1995; Arenas *et al*. 2000). Annual nesting at the principal beach, Escobilla Beach, Oaxaca, Mexico, averaged 138,000 nests prior to the ban, and since the ban on harvest in 1990, annual nesting has increased to an average of 525,000 nests (Salazar *et al.* in press). At a smaller olive ridley nesting beach in central Mexico, Playon de Mismalayo, nest and egg protection efforts have resulted in more hatchlings, but the population is still seriously decremented and is threatened with extinction (Silva-Batiz *et al*. 1996). Nevertheless some authors have suggested that olive ridley turtles in Mexico should be considered recovered (Arenas *et al*. 2000).

The main threats to turtles in Thailand include egg poaching, harvest and subsequent consumption or trade of adults or their parts (i.e. carapace), indirect capture in fishing gear, and loss of nesting beaches through development (Aureggi *et al*. 1999). During the 1996-97 survey, only six olive ridley nests were recorded, and of these, half were poached, and one was predated by feral dogs. During the 1997-98 survey, only three nests were recorded.

Olive ridley nests in Indonesia are subject to extensive hunting and egg collection. In combination with rapid rural and urban development, these activities have reduced the size of the nesting population in the region as well as their nesting success.

Atlantic Salmon (Gulf of Maine Population)

Species description and distribution

Atlantic salmon are an anadromous species: spawning and juvenile rearing occur in freshwater rivers followed by migration to the marine environment. This listing includes wild Atlantic salmon found in rivers and streams from the lower Kennebec River north to the border between the U.S. and Canada, including the Dennys, East Machias, Machias, Pleasant, Narraguagus, Ducktrap, and Sheepscot Rivers and Cove Brook. While at sea, Atlantic salmon undertake extensive migrations to waters off Canada and Greenland. Data from past commercial harvest indicate that post-smolts overwinter in the southern Labrador Sea and in the Bay of Fundy.

Life history information

Juvenile salmon in New England rivers typically migrate to sea in May after a two to three year period of development in freshwater streams, and remain at sea for two winters before returning to their U.S. natal rivers to spawn from mid October through early November. After migrating to salt water, Atlantic salmon form small schools and loose aggregations that swim close to the ocean's surface of nearshore environments where they feed on invertebrates, amphipods, euphausiids, and fish (Biological Review Team 1999).

During the first winter, some of these fish overwinter in the Bay of Fundy. After the first winter, Atlantic salmon become more solitary and feed primarily on capelin, herring, and sand lance. Although these Atlantic salmon appear to migrate in the upper six meters of the ocean, information on their marine predators suggest that they occupy deeper water during some portion of their time at sea (Biological Review Team 1999).

Listing status

The Gulf of Maine population of Atlantic salmon was listed as endangered under the Endangered Species Act of 1973 on December 18, 2000.

Population status and trends

The abundance of wild, Gulf of Maine Atlantic salmon is perilously small: the total run size of spawning adults in this species numbered approximately 150 animals in 1999 (Baum 2000). Since 1992, no wild Atlantic salmon have been caught in commercial fisheries or by research or survey vessels within the distribution of this species.

Impacts of human activities on this species

This population of Atlantic salmon is threatened by a combination of habitat modification and destruction, historical overharvests in commercial and recreational fisheries, disease and predation, and natural environmental change. Habitat for this species has been destroyed or adversely affected by (1) water extraction; (2) sedimentation; (3) obstructions to passage including those caused by beaver and debris dams and poorly designed road crossings; (4) input of nutrients; (5) chronic exposure to insecticides, herbicides, fungicides, and pesticides (in particular, those used to control spruce budworm); (6) elevated water temperatures from processing water discharges; and (7) removal of vegetation along streambanks. The most obvious and immediate threat is posed by water extraction on some rivers within the DPS range, as it has the potential to expose or reduce salmon habitat.

Both commercial and recreational harvest of Atlantic salmon historically played a role in the decline of wild, Gulf of Maine Atlantic salmon. Atlantic salmon parr remain vulnerable to harvest by trout anglers, and mortality associated with this activity has been documented. Fish diseases have always represented a source of mortality to Atlantic salmon in the wild, although the threat of major epizootics have been generally associated with salmon aquaculture. Finally, predation has influenced the size of Atlantic salmon population; this species' predators include cormorants, striped bass, and several species of seals.

Chinook Salmon

Chinook salmon are the largest of the Pacific salmon and historically ranged from the Ventura River in California to Point Hope, Alaska in North America, and in northeastern Asia from Hokkaido, Japan to the Anadyr River in Russia (Healey 1991). In addition, chinook salmon have been reported in the Canadian Beaufort Sea (McPhail and Lindsey 1970). Because of similarities in the life history and threats to the survival and recovery of the six chinook salmon evolutionary significant units (ESU) that are included in this biological opinion, we will summarize the general life history and threats to chinook salmon generally. Then we will separately discuss specific information on their listing status, population status and trends, and impacts that are not shared for each ESU.

Ocean Distribution and Abundance

Chinook salmon distribute in the North Pacific Ocean north of about 40º North latitude where they may remain for 1 to 6 years, although 2 to 4 years are more common. Although salmon generally occur near the surface (within 8 to 10 meters of the surface), chinook salmon have been caught at depths up to 110 meters.

Life history information

Chinook salmon exhibit diverse and complex life history strategies. Two generalized freshwater life-history types were initially described by Gilbert (1912): "stream-type" chinook salmon reside in freshwater for a year or more following emergence, whereas "ocean-type" chinook salmon migrate to the ocean within their first year.

The generalized life history of chinook salmon involves incubation, hatching, and emergence in freshwater, migration to the ocean, and subsequent initiation of maturation and return to freshwater for completion of maturation and spawning. Juvenile rearing in freshwater can be minimal or extended. Additionally, some male chinook salmon mature in freshwater, thereby foregoing emigration to the ocean.

Impacts of human activity on chinook salmon

Over the past few decades, the size and distribution of chinook salmon populations have declined because of natural phenomena and human activity, including the operation of hydropower systems, over-harvest, hatcheries, and habitat degradation. Natural variations in freshwater and marine environments have substantial effects on the abundance of salmon populations. Of the various natural phenomena that affect most populations of Pacific salmon, changes in ocean productivity are generally considered most important.

Chinook salmon are exposed to high rates of natural predation, particularly during freshwater rearing and migration stages. Ocean predation probably contributes to significant natural mortality, although the levels of predation are largely unknown. In general, chinook are prey for pelagic fishes, birds, and marine mammals, including harbor seals, sea lions, and killer whales. There have been recent concerns that the increasing size of tern, seal, and sea lion populations in the Pacific Northwest has dramatically reduced the survival of adult and juvenile salmon.

Puget Sound chinook salmon

Species description and distribution

Puget Sound chinook salmon include all runs of chinook salmon in the Puget Sound region from the North Fork Nooksack River to the Elwha River on the Olympic Peninsula. Chinook salmon in this area generally have an “ocean-type” life history. Thirty-six hatchery populations were included as part of the ESU and five were considered essential for recovery and listed including spring chinook from Kendall Creek, the North Fork Stillaguamish River, White River, and Dungeness River, and fall run fish from the Elwha River.

Listing status

Puget Sound chinook salmon were listed as threatened under the ESA in 1999. Critical habitat designated for this ESU was vacated on May 7, 2002.

Population status and trends

The largest recorded harvest of this species occurred in 1908, when the run-size for Puget Sound chinook salmon was estimated at 690,000 fish (in 1908, both ocean harvests and hatchery production were negligible). Between 1992 and 1996, the average run-size of natural chinook salmon runs in North Puget Sound was about 13,000 fish. With few exceptions, these runs represented short- and long-term declines.

Lower Columbia River chinook salmon

Species description and distribution

Lower Columbia River chinook salmon includes all native populations from the mouth of the Columbia River to the crest of the Cascade Range, excluding populations above Willamette Falls. The Cowlitz, Kalama, Lewis, White Salmon, and Klickitat Rivers are the major river systems on the Washington side, and the lower Willamette and Sandy Rivers are foremost on the Oregon side. The eastern boundary for this species occurs at Celilo Falls, which corresponds to the edge of the drier Columbia Basin Ecosystem and historically may have been a barrier to salmon migration at certain times of the year.

Fall-run fish form the majority of these chinook salmon, whose stocks tend to migrate north once they reach the ocean. This is supported by recoveries of coded-wire-tags for lower Columbia

River chinook salmon, which tend to be recovered off the British Columbia and Washington coasts, with a small proportion recovered in Alaskan waters.

Stream-type spring-run chinook salmon found in the Klickitat River are not included in this species (they are considered Mid-Columbia River spring-run chinook salmon) or the introduced Carson spring-chinook salmon strain. "Tule" fall chinook salmon in the Wind and Little White Salmon Rivers are included in this ESU, but not introduced "upriver bright" fall-chinook salmon populations in the Wind, White Salmon, and Klickitat Rivers.

There is some question whether any natural-origin spring chinook salmon remain in this species. Fourteen hatchery stocks were included in the ESU; one was considered essential for recovery (Cowlitz River spring chinook) but was not listed.

Listing status

Lower Columbia River chinook salmon were listed as threatened under the ESA in 1999. Critical habitat designated for this ESU was vacated on May 7, 2002.

Population status and trends

There are no reliable estimates of the historic abundance of Lower Columbia River chinook salmon, but experts generally agree that naturally-spawning populations of this species have declined dramatically over the last century. By the 1990s, spawning runs of this species have been sustained by hatchery production. For example, between 1991 and 1995, estimated escapements of this species have included 29,000 natural spawners and 37,000 hatchery spawners and about 68% of the natural spawners were first-generation hatchery strays (PFMC 1996).

Upper Columbia River spring-run chinook salmon

Species description and distribution

Upper Columbia River spring-run chinook salmon ESU includes stream-type chinook salmon that inhabit tributaries upstream from the Yakima River to Chief Joseph Dam. They currently spawn in only three river basins above Rock Island Dam: the Wenatchee, Entiat, and Methow Rivers. Several hatchery populations are also listed including those from the Chiwawa, Methow, Twisp, Chewuch, and White rivers, and Nason Creek.

Adults of this species return to the Wenatchee River from late March to early May, and from late March to June in the Entiat and Methow rivers. Most adults return after spending two years in the ocean, while 20%-40% return after three years at sea. Like the Snake River spring/summer-run chinook, Upper Columbia River spring-run chinook are subject to very little ocean harvest.

Listing status

Upper Columbia River spring-run chinook salmon were listed as endangered under the ESA in 1999. Critical habitat designated for this ESU was vacated on May 7, 2002.

Population status and trends

There are no historical estimates of the size of Upper Columbia chinook salmon populations. Adult escapements of this species throughout its range continue to be critically low and redd counts are still declining severely.

Upper Columbia River chinook salmon have been reduced to small populations in three watersheds. Population viability analyses for this species (using the Dennis Model) suggest that these chinook salmon face a significant risk of extinction: a 75 to 100 percent probability of extinction within 100 years (given return rates for 1980 to present).

Upper Willamette River Chinook Salmon

Species description and distribution

Upper Willamette River chinook salmon occupy the Willamette River and tributaries upstream of Willamette Falls. Historically, access above Willamette Falls was restricted to the spring when flows were high. In autumn, low flows prevented fish from ascending past the falls. The Upper Willamette spring-run chinook are one of the most genetically distinct chinook groups in the Columbia River Basin. Fall-run chinook salmon spawn in the Upper Willamette but are not considered part of the species because they are not native. None of the hatchery populations in the Willamette River were listed although five spring-run hatchery stocks were included in the ESU.

The ocean distribution of Upper Willamette River chinook salmon is consistent with an ocean-type life history with the majority of chinook being caught off the coasts of British Columbia and Alaska. Spring chinook from the Willamette River have the earliest return timing of chinook stocks in the Columbia Basin with freshwater entry beginning in February. Historically, spawning occurred between mid-July and late October. However, the current spawn timing of hatchery and wild chinook in September and early October has probably been changed through introgression with hatchery salmon.

Listing status

Upper Willamette River chinook salmon were listed as threatened under the ESA in 1999. Critical habitat designated for this ESU was vacated on May 7, 2002.

Population status and trends

Populations of naturally produced Upper Willamette River spring chinook are substantially smaller than they were historically, when escapement levels may have been as high as 200,000 fish per year. The Willamette River's ability to produce salmon has been reduced by extensive dam construction and habitat degradation. In response, chinook salmon populations in the Willamette River have declined. From 1946 to 1950, geometric mean counts of spring chinook was 31,000 fish, primarily naturally-produced salmon (Myers *et al*. 1998). From 1995 to 1999, geometric mean counts of spring chinook salmon was 27,800 fish, primarily hatchery-produced salmon.

Snake River Spring/summer-run Chinook Salmon

Species description and distribution

Snake River spring/summer-run chinook salmon are primarily limited to the Salmon, Grande Ronde, Imnaha, and Tucannon Rivers in the Snake River basin. Most adult Snake River spring/summer-run chinook salmon enter these rivers to spawn from May through September. Juvenile Snake River spring/summer-run chinook salmon emerge from spawning gravels from February through June. After rearing in nursery streams for about one year, smolts begin migrating seaward in April and May. After reaching the mouth of the Columbia River, spring/summer-run chinook salmon probably inhabit nearshore areas before migrating to the northeast Pacific Ocean where they will remain for two to three years.

Listing status

Snake River spring/summer-run chinook salmon were listed as endangered under the ESA in 1992. Critical habitat for these salmon was designated in 1993. This critical habitat encompasses the waters, waterway bottoms, and adjacent riparian zones of specified lakes and river reaches in the Columbia River that are or were accessible to listed Snake River salmon (except reaches above impassable natural falls, and Dworshak and Hells Canyon Dams) and is well beyond the area that is likely to be directly or indirectly affected by the proposed action.

Population status and trends

In the late 1800s, the population of wild, adult Snake River spring/summer-run chinook salmon was estimated at more than 1.5 million adults. By the 1950s, the population had declined to an estimated 125,000 adults and continued to decline through the 1970s. Returns were variable through the 1980s, but declined further in the 1990s. Record low returns were observed in 1994 and 1995. Dam counts were modestly higher from 1996-1998, but declined in 1999.

In 2000, 134,000 Snake River spring-run chinook salmon were expected to return to the Snake River, which would be the highest return in over 30 years. Only a small portion of these returning salmon (5,800) are expected to be natural-origin spring chinook destined for the Snake River. Expected returns to the Tucannon River (500 listed hatchery and wild fish), Imnaha River (800 wild and 1,600 listed hatchery fish), and Sawtooth Hatchery (368 listed hatchery fish) all represent substantial increases over past years.

In 2000, 33,300 Snake River summer-run chinook salmon were expected to return to the Snake River, which is the second highest return in over 30 years, but only a small portion of these animals (2,000) are expected to be natural-origin salmon. The return of natural-origin fish is slightly more than half of the five-year average (3,466).

In 1999, NMFS conducted an analysis referred to as Cumulative Risk Initiative, which estimated the Snake River spring/summer-run chinook salmon's probability of extinction for 10- and 100-year periods (NWFSC 1999). For some of the index stocks of this species, the risk analysis estimated the Marsh River subpopulation had a 90 percent probability of extinction within 100 years; the Imnaha River subpopulation had a 74 percent probability of extinction within 100 years; the Bear Creek and Sulphur River subpopulations had 50 percent probabilities of extinction; and the remaining three subpopulations had extinction probabilities that ranged between 30 and 40 percent.

The Northwest Fisheries Science Center has recently considered the extinction risk for Snake River spring/summer-run chinook as part of their Cumulative Risk Initiative, which was based on seven "index" populations of Snake River spring/summer-run chinook salmon (out of a total of 35 to 40 populations). Two populations have a 10 percent risk of declining to one individual in ten years, four populations have 56 to 88 percent probability of declining to one individual in 100 years that range between 56 and 88 percent, and the remaining three populations have more than 30 percent probability of declining to this level within 100 years if nothing changes.

Snake River fall-run chinook salmon

Species description and distribution

The present range of spawning and rearing habitat for naturally-spawned Snake River fall chinook salmon is primarily limited to the Snake River below Hells Canyon Dam and the lower reaches of the Clearwater, Grand Ronde, Salmon, and Tucannon Rivers.

Life history information

Unlike many other listed salmon, Snake River fall-run chinook are probably represented by only a single population that spawns in parts of the mainstem of the river and lower reaches of tributaries. Adult Snake River fall-run chinook salmon enter the Columbia River in July and migrate into the Snake River from August through October. Fall chinook salmon generally spawn from October through November and fry emerge from March through April. Downstream migration generally begins within several weeks of emergence (Becker 1970, Allen and Meekin 1973), and juveniles rear in backwaters and shallow water areas through mid-summer prior to smolting and migrating to the ocean, thus they exhibit an "ocean" type juvenile history. Once in the ocean, they spend one to four years (usually three) before beginning their spawning migration. Fall returns in the Snake River system are typically dominated by four-year-old fish.

Listing status

Snake River fall-run chinook salmon were listed as endangered under the ESA in 1992. Critical habitat for these salmon was designated in 1993. This critical habitat encompasses the waters, waterway bottoms, and adjacent riparian zones of specified lakes and river reaches in the Columbia River that are or were accessible to listed Snake River salmon (except reaches above impassable natural falls, and Dworshak and Hells Canyon Dams) and is well beyond the area that is likely to be directly or indirectly affected by the proposed action.

Population status and trends

There are no reliable estimates of historical population sizes of Snake River fall chinook salmon. The mean number of adult Snake River fall chinook salmon was estimated to have declined from 72,000 in the 1930s and 1940s to 29,000 during the 1950s. In spite of these declines, the Snake River was the most important area of natural production of fall chinook in the Columbia River basin through the 1950s. The number of adults counted at the uppermost Snake River mainstem dams averaged 12,720 total spawners from 1964 to 1968, 3,416 spawners from 1969 to 1974, and 610 spawners from 1975 to 1980 (Waples, *et al*. 1991). Counts of adult fish of natural-origin continued to decline through the 1980s when they reached a low of 78 individuals in 1990. Since 1990, returns of natural-origin fish to Lower Granite Dam have been variable, but increasing. They reached a high of 797 in 1997 only to decline to 306 in 1998.

The Lyons Ferry Hatchery populations of Snake River fall-run chinook, which was included in this species' listing helps buffer this species from natural declines. In recent years, several hundred adult fall chinook salmon have returned to Lyons Ferry Hatchery and smolt from the 1995 brood-year were outplanted to accelerate rebuilding this species. Nevertheless, supplementation will not substitute for habitat restoration to recover this species because of this species' ecology.

The Northwest Fisheries Science Center recently considered the extinction risk for Snake River fall-run chinook as part of their Cumulative Risk Initiative. The results of these analyses indicate that the probability of extinction for Snake River fall chinook over the next ten years is near zero while the risk of extinction over 100 years is between 6-17% (depending on whether 1980 is included in the baseline analysis).

Sacramento River winter-run chinook salmon

Species description and distribution

The Sacramento River winter-run chinook salmon ESU includes populations of winter-run chinook salmon in the Sacramento River and its tributaries in California. Winter-run chinook salmon generally have an "ocean-type" life history. The Sacramento River winter-run chinook ESU consists of a single spawning population that enters the Sacramento River from November to June and spawns from late April to mid-August, with a peak from May to June.

Listing status

Sacramento River winter-run chinook salmon were listed as endangered under the ESA in 1994. Critical Habitat was designated in 1993. The critical habitat for this ESU is well beyond the area that is likely to be directly or indirectly affected by the proposed action.

Population status and trends

Sacramento River winter chinook are listed as endangered because they presently have access to a small fraction of their historic spawning habitat and the habitat remaining to them is degraded. In addition, they face hostile downstream conditions in the mainstem Sacramento River and the Sacramento-San Joaquin Delta; they are caught in ocean and, occasionally, in freshwater fisheries. Between 1970 and 1990, the spawning population declined from over 50,000 fish to less than a thousand.

Sacramento River winter chinook historically occupied cold, headwater streams, such as the upper reaches of the Little Sacramento, McCloud, and lower Pit Rivers. The ESU presently consists of a single self-sustaining population which is entirely dependent upon the provision of suitably cool water from Shasta Reservoir during periods of spawning, incubation and rearing.

Since 1993, the winter chinook population has increased in abundance. Several estimates of spawner abundance are now available. The factors responsible for the increase likely include improvements in fresh water spawning and migration habitat, as well as reductions in ocean harvest of the population. The mean 3-year adult replacement rate over the past three years, based on the Petersen carcass survey estimates, is 2.6.

Central Valley Spring-run chinook salmon

Species description and distribution

The Central Valley Spring-run chinook salmon ESU includes all naturally spawned populations of spring-run chinook salmon in the Sacramento River and its tributaries in California. This ESU includes chinook salmon entering the Sacramento River from March to July and spawning from late August through early October, with a peak in September. Spring-run fish in the Sacramento River exhibit an ocean-type life history, emigrating as fry, subyearlings, and yearlings.

Listing status

Central Valley Spring-run chinook salmon were listed as threatened under the ESA in 1999. The Critical Habitat designation for this ESU was vacated on May 7, 2002.

Population status and trends

Spring chinook are listed as threatened because they presently have access to a small fraction (perhaps 10% or less) of their historic spawning habitat and the habitat remaining to them is degraded. In addition, they face hostile downstream conditions in the mainstem Sacramento River and the Sacramento-San Joaquin Delta, they are caught in ocean and freshwater fisheries and they may be subject to the adverse genetic affects of straying hatchery populations such as Feather River Hatchery spring chinook.

Spring chinook historically occupied the upper reaches of all major tributaries to the Sacramento and San Joaquin rivers. Of the 21 populations identified by the California Department of Fish and Game in their status review, only 3 self-sustaining populations now exist in the upper Sacramento in Deer, Mill and Butte Creeks. Although these streams have not been affected by large impassable dams, diversions and small dams have degraded the spawning habitat.

Since 1993, spring chinook populations have increased in abundance. The factors responsible for these increases likely include adequate rainfall, improvements in fresh water spawning and migration habitat, as well as the reduction in harvest rates on Central Valley chinook during the last three years. Efforts to restore salmon habitat in Butte Creek have been underway for the past decade. Over 20,000 spring chinook returned to Butte Creek in 1998 and 3,000 in 1999. Both years represent greater than two fold increases in the three-year replacement rate.

California Coastal chinook salmon

Species description and distribution

The California Coastal chinook salmon ESU includes all naturally spawned coastal chinook salmon spawning from Redwood Creek south through the Russian River, inclusive.

Listing status

California Coastal chinook salmon were listed as threatened under the ESA in 1999. The Critical Habitat designation for this ESU was vacated on May 7, 2002.

Population status and trends

California coastal chinook are listed as threatened as a result of habitat blockages, logging, agricultural activities, urbanization, and water withdrawals in the river drainages that support California coastal salmon. These have resulted in widespread declines in abundance of chinook relative to historical levels and the present distribution of small populations with sporadic occurrences. Smaller coastal drainages such as the Noyo, Garcia and Gualala rivers may have supported chinook salmon runs historically, but they contain few or no fish today. The Russian River probably contains some natural production, but the origin of those fish is not clear because of a number of introductions of hatchery fish over the last century. The Eel River contains a substantial fraction of the remaining chinook salmon spawning habitat within the ESU. Where available, surveys of coastal chinook spawner abundance in some cases show improvement relative to the extremely low escapements of the early 90s; other streams, such as Tomki Creek remain extremely depressed.

Chum Salmon

Historically, chum salmon were distributed throughout the coastal regions of western Canada and the United States, as far south as Monterey Bay, California. Presently, major spawning populations are found only as far south as Tillamook Bay on the northern Oregon coast. Chum salmon are semelparous, spawn primarily in freshwater and, apparently, exhibit obligatory anadromy (there are no recorded landlocked or naturalized freshwater populations) (Randall *et al.* 1987).

Chum salmon spend two to five years in feeding areas in the northeast Pacific Ocean, which is a greater proportion of their life history than other Pacific salmonids. Chum salmon distribute throughout the North Pacific Ocean and Bering Sea, although North American chum salmon (as opposed to chum salmon originating in Asia), rarely occur west of 175° E longitude (Johnson *et al.* 1997).

North American chum salmon migrate north along the coast in a narrow coastal band that broadens in southeastern Alaska, although some data suggest that Puget Sound chum, including Hood Canal summer run chum, may not make extended migrations into northern British Columbian and Alaskan waters, but instead may travel directly offshore into the north Pacific Ocean (Johnson *et al.* 1997).

Chum salmon, like pink salmon, usually spawn in the lower reaches of rivers, with redds usually dug in the mainstem or in side channels of rivers from just above tidal influence to nearly 100 km from the sea. Juveniles outmigrate to seawater almost immediately after emerging from the gravel that covers their redds (Salo 1991). This ocean-type migratory behavior contrasts with the stream-type behavior of some other species in the genus Oncorhynchus (e.g., coastal cutthroat trout, steelhead, coho salmon, and most types of chinook and sockeye salmon), which usually migrate to sea at a larger size, after months or years of freshwater rearing. This means that survival and growth in juvenile chum salmon depend less on freshwater conditions (unlike stream-type salmonids which depend heavily on freshwater habitats) than on favorable estuarine conditions. Another behavioral difference between chum salmon and species that rear extensively in freshwater is that chum salmon form schools, presumably to reduce predation (Pitcher 1986), especially if their movements are synchronized to swamp predators (Miller and Brannon 1982).

Chum salmon have been threatened by overharvests in commercial and recreational fisheries, adult and juvenile mortalities associated with hydropower systems, habitat degradation from forestry and urban expansion, and shifts in climatic conditions that changed patterns and intensity of precipitation.

Columbia River Chum Salmon

Species description and distribution

Columbia River chum salmon includes all natural-origin chum salmon in the Columbia River and its tributaries in Washington and Oregon. The species consists of three populations: Grays River, Hardy, and Hamilton Creek in Washington State

Listing status

Columbia River chum salmon were listed as threatened on March 25, 1999 (64 FR 14508). Critical habitat was designated for Columbia River chum salmon on February 16, 2000 (65 FR 7764), but the designation for this ESU was vacated on May 7, 2002.

Population status and trends

Columbia River chum salmon abundance is probably less than 1% of historical levels, and the species has lost some (perhaps much) of its original genetic diversity. Average annual natural escapement to the index spawning areas was approximately 1,300 fish from 1990 through 1998 (Johnson *et al.* 1997).

Hood Canal Summer-run Chum Salmon

Species description and distribution

Hood Canal summer-run chum salmon includes summer-run chum salmon populations in Hood Canal in Puget Sound and in Discovery and Sequim Bays on the Strait of Juan de Fuca. It may also include summer-run fish in the Dungeness River, but the existence of that run is uncertain. Five hatchery populations are considered part of the ESU including those from the Quilcene National Fish Hatchery, Long Live the Kings Enhancement Project (Lilliwaup Creek), Hamma Hamma River Supplementation Project, Big Beef Creek reintroduction Project, and the Salmon Creek supplementation project in Discovery Bay. Although included as part of the ESU, none of the hatchery populations were listed.

Listing status

Hood Canal summer-run chum salmon were listed as endangered under the ESA on March 25, 1999. Critical habitat designated for this ESU was vacated on May 7, 2002.

Population status and trends

Of the sixteen populations of summer chum that are included in this species, seven are considered to be "functionally extinct" (Skokomish, Finch Creek, Anderson Creek, Dewatto, Tahuya, Big Beef Creek, and Chimicum). The remaining nine populations are well distributed throughout the range of the species except for the eastern side of Hood Canal; those populations are among the least productive (Johnson *et al.* 1997).

Coho Salmon

Coho salmon occur naturally in most major river basins around the North Pacific Ocean from central California to northern Japan (Laufle *et al.* 1986). After entering the ocean, immature coho salmon initially remain in near-shore waters close to the parent stream. Most coho salmon adults are 3-year-olds, having spent approximately 18 months in freshwater and 18 months in salt water. Wild female coho return to spawn almost exclusively at age 3. Spawning escapements of coho salmon are dominated by a single year class. The abundance of year classes can fluctuate dramatically with combinations of natural and human-caused environmental variation.

North American coho salmon will migrate north along the coast in a narrow coastal band that broadens in southeastern Alaska. During this migration, juvenile coho salmon tend to occur in both coastal and offshore waters. During spring and summer, coho salmon will forage in waters between 46° N, the Gulf of Alaska, and along Alaska's Aleutian Islands.

The factors threatening naturally reproducing coho salmon throughout its range are numerous and varied. For coho salmon populations in California and Oregon, the present depressed condition is the result of several longstanding, human-induced factors (e.g., habitat degradation, water diversions, harvest, and artificial propagation) that serve to exacerbate the adverse effects of natural environmental variability from such factors as drought, floods, and poor ocean conditions. The major activities responsible for the decline of coho salmon in Oregon and California are logging, road building, grazing, mining activities, urbanization, stream channelization, dams, wetland loss, water withdrawals and unscreened diversions for irrigation.

Central California Coho Salmon

Species description and distribution

Central California coho salmon consist of all coho salmon that reproduce in streams between Punta Gorda and the San Lorenzo River, including hatchery stocks (except for the Warm Springs Hatchery on the Russian River), although hatchery populations are not listed.

Listing status

Critical habitat for central California coho salmon encompasses accessible reaches of all rivers (including estuarine areas and tributaries) between Punta Gorda and the San Lorenzo River, and Mill Valley and Corte Madera Creek which enter the San Francisco Bay (62 FR 62741, November 25, 1997).

Population status and trends

Of 186 streams in the range of central California coho salmon identified as having historic accounts of adult coho salmon, recent data exist for 133 (72 %). Of these 133 streams, 62 (47 %) have recent records of occurrence of adult coho salmon and 71 (53 %) no longer maintain coho salmon spawning runs (Brown *et al.* 1994).

Oregon Coastal Coho Salmon

Species description and distribution

Oregon coastal coho salmon include naturally spawning populations of coho salmon inhabiting coastal streams between Cape Blanco, Oregon, and the Columbia River.

Listing status

After reviewing biological data on the species' status and an assessment of protective efforts, NMFS concluded in August 1997 that this ESU did not warrant listing. However, the Oregon District Court overturned the decision and NMFS listed the ESU as threatened on August 10, 1998. Critical habitat designated for this ESU was vacated on May 7, 2002.

Population status and trends

Based on historic commercial landing statistics and estimated exploitation rates, coho salmon escapement to coastal Oregon rivers has been estimated at between 1 and 1.4 million fish in the early 1900s with harvest of nearly 400,000 fish (Mullen 1981; Lichatowich 1989). Recent spawning escapement from 1991-1993 has been estimated at an annual average of about 39,000 adults (Jacobs and Cooney 1991, 1992, 1993). This decline has been associated with a reduction in habitat capacity of nearly 50% (Lichatowich 1989). Current production potential for coho salmon in coastal Oregon rivers has been estimated at about 800,000 fish using stock-recruit models (Lichatowich 1989).

Southern Oregon/Northern California Coast Coho Salmon

Species description and distribution

Southern Oregon/Northern California coasts coho salmon (SONCC) consists of all naturally spawning populations of coho salmon that reside below long-term, naturally impassible barriers in streams between Punta Gorda, California and Cape Blanco, Oregon. The geographic area of the listed species encompasses five of the seven hatchery stocks reared and released within the species' range of the species although none of the hatchery populations are listed.

The three major river systems supporting coho in the SONCC ESU are the Rogue, Klamath (including the Trinity), and Eel rivers.

Listing status

SONCC salmon were listed as threatened on August 18, 1997. Critical habitat for the ESU encompasses accessible reaches of all rivers (including estuarine areas and tributaries) between the Mattole River in California and the Elk River in Oregon, inclusive (62 FR 62741, November 25, 1997).

Population status and trends

Of the 396 streams within the range of the California portion of the SONCC ESU that were identified as once having coho salmon runs, recent survey information is available for 115 streams (29 percent). Of these 115 streams, 73 (64 percent) still support coho salmon runs while 42 (36 percent) have lost their coho salmon runs. The rivers and tributaries in the California portion of the SONCC ESU were estimated to have average recent run sizes of 7,080 natural spawners and 17,156 hatchery returns, with 4,480 identified as native fish occurring in tributaries having little history of supplementation with non-native fish (Brown *et al.* 1994).

Sockeye Salmon

Sockeye salmon occur in the North Pacific and Arctic oceans and associated freshwater systems. This species ranges south as far as the Klamath River in California and northern Hokkaido in Japan, to as far north as far as Bathurst Inlet in the Canadian Arctic and the Anadyr River in Siberia. Sockeye salmon were an important food source for aboriginal people who either ate them fresh or dried them for winter use. Today sockeye salmon remain an important mainstay of many subsistence users and support one of the most important commercial and recreational fisheries on the Pacific coast of North America.

Sockeye salmon can be distinguished from chinook, coho, and pink salmon by the lack of large, black spots and from chum salmon by the number and shape of gill rakers on the first gill arch. Sockeye salmon have 28 to 40 long, slender, rough or serrated closely set rakers on the first arch. Chum salmon have 19 to 26 short, stout, smooth rakers.

Snake River Sockeye Salmon

Species description and distribution

Snake River sockeye salmon are one of three stocks of sockeye salmon remaining in the Columbia River basin. This species includes sockeye populations from the Snake River Basin, Idaho, although the only remaining populations of this species occur in the Stanley River Basin of Idaho.

Life history information

Adult Snake River sockeye salmon enter the Columbia River during June and July. Their arrival at Redfish Lake, which now supports the only remaining run of Snake River sockeye salmon, peaks in August; spawning occurs primarily in October. Eggs hatch in the spring between 80 and 140 days after spawning. Fry remain in the gravel for three to five weeks, emerge from April through May and move immediately into the lake. Once there, juvenile sockeye salmon feed on plankton for one to three years before they migrate to the ocean. Migrants leave Redfish Lake from late April through May and smolts migrate almost 900 miles to the Pacific Ocean.

Smolts pass Lower Granite Dam (the first dam on the Snake River downstream from the Salmon River) from late April to July with peak passage from May to late June (Fish Passage Center 1992). Once in the ocean, Snake River sockeye salmon smolts remain inshore or within the Columbia River influence during the early summer. Later, they migrate through the northeast Pacific Ocean where they remain for two to three years (Hart 1973, Hart and Dell 1986). Snake River sockeye salmon usually begin the spawning migration in their fourth or fifth year of life.

Listing status

Snake River sockeye salmon were listed as endangered under the ESA in 1991. Critical habitat for these salmon was designated in 1993. This critical habitat encompasses the waters, waterway bottoms, and adjacent riparian zones of specified lakes and river reaches in the Columbia River that are or were accessible to listed Snake River salmon (except reaches above impassable natural falls, and Dworshak and Hells Canyon Dams) and is well beyond the area that is likely to be affected by the proposed action.

Population status and trends

Historically, the largest numbers of Snake River sockeye salmon returned to headwaters of the Payette River, where 75,000 were taken in one year by a single fishing operation on Big Payette Lake (Bevan *et al*. 1994). During the early 1880s, returns of Snake River sockeye salmon to the headwaters of the Grande Ronde River in Oregon were estimated between 24,000 and 30,000 at a minimum. During the 1950s and 1960s, adult returns to Redfish Lake numbered more than 4,000 fish. By 1985, the number of adults arriving at Redfish Lake, Idaho, had fallen below 20 animals. Between 1990 and 1998, only 16 “wild” Snake River sockeye salmon returned to Redfish Lake or the nearby Sawtooth Hatchery (including one in 1998 and none in 1999).

Since 1991, all returning adults Snake River sockeye salmon have been spawned in a hatchery to prevent the species’ extinction. The first adults produced by this program (from the 1991 returns) were released into Redfish Lake to spawn in 1993 and their progeny were expected to outmigrate in the spring of 1995. Sixteen sockeye were observed at Lower Granite Dam in 1999, seven of which return to the Sawtooth Hatchery weir. By Aug. 8 of 2000, 149 four-year-old sockeye adults had made the 900-mile journey from the ocean to Redfish Lake or Sawtooth Hatchery. Most are products of either sockeye adults produced in the hatchery program and released to spawn in 1996 or year-old smolts released near the hatchery or in Redfish Creek. All are progeny of eight, lone returning "wild" sockeye salmon that had been taken into the program as broodstock in 1993.

Given the extremely low sockeye salmon population size, this species' likelihood of surviving in the wild remains fairly low. Snake River sockeye will remain below the threshold escapement level of 150 fish (which applies only to naturally-produced spawners) until natural production is sufficiently re-established. This species' likelihood of recovering in the wild (which only applies to spawners at least two generations removed from captive broodstock) is even less certain.

Steelhead

Five threatened or endangered ESUs of steelhead are known to occur in the action area for this consultation. Unlike Pacific salmon, steelhead are capable of spawning more than once before death (iteroparity). However, steelhead rarely spawn more than twice before dying; most that do so are females (August 9, 1996, 61 FR 41542). Biologically, steelhead can be divided into two basic run-types: the stream-maturing type, or summer steelhead, enters fresh water in a sexually immature condition and requires several months in freshwater to mature and spawn and the ocean-maturing type, or winter steelhead, enters fresh water with well-developed gonads and spawns shortly after river entry (August 9, 1996, 61 FR 41542; Burgner *et al*. 1992). Variations in migration timing exist between populations. Some river basins have both summer and winter steelhead, while others only have one run-type.

Ocean Distribution and Abundance

The ocean distributions for listed steelhead are not known in detail, but steelhead are caught only rarely in ocean salmon fisheries. The total catch of steelhead in Canadian fisheries is low and consideration of the likely stock composition suggests that the catch of listed steelhead is less than 10 per year from the five steelhead ESUs combined (NMFS 1999a).

General life history information

Summer steelhead enter freshwater between May and October in the Pacific Northwest (Busby *et al*. 1996). They require cool, deep holding pools during summer and fall, prior to spawning. They migrate inland toward spawning areas, overwinter in the larger rivers, resume migration in early spring to natal streams, and then spawn (Meehan and Bjornn 1991).

Winter steelheads enter freshwater between November and April in the Pacific Northwest (Busby *et al*. 1996), migrate to spawning areas, and then spawn in late winter or spring. Some adults, however, do not enter coastal streams until spring, just before spawning. Steelhead typically spawn between December and June (Bell 1991), and the timing of spawning overlaps between populations regardless of run type (Busby *et al*. 1996).

Steelhead spawn in cool, clear streams featuring suitable gravel size, depth, and current velocity. Intermittent streams may also be used for spawning (Barnhart 1986; Everest 1973). Depending on water temperature, steelhead eggs may incubate for 1.5 to 4 months (August 9, 1996, 61 FR 41542) before hatching. Juveniles rear in fresh water from one to four years, then migrate to the ocean as smolts (August 9, 1996, 61 FR 41542). Winter steelhead populations generally smolt after two years in fresh water (Busby *et al*. 1996).

Steelhead typically reside in marine waters for two or three years before migrating to their natal streams to spawn as four- or five-year olds (August 9, 1996, 61 FR 41542). Populations in Oregon and California have higher frequencies of age-1-ocean steelhead than populations to the north, but age-2-ocean steelhead generally remain dominant (Busby *et al*. 1996). Age structure appears to be similar to other west coast steelhead, dominated by four-year-old spawners (Busby *et al*. 1996).

Upper Columbia River Steelhead

Species description and distribution

Upper Columbia River steelhead occupy the Columbia River Basin upstream from the Yakima River, Washington, to the border between the United States and Canada. This area includes the Wenatchee, Entiat, and Okanogan Rivers. All upper Columbia River steelhead are summer steelhead. Steelhead primarily use streams of this region that drain the northern Cascade Mountains of Washington State. This species includes hatchery populations of summer steelhead from the Wells Hatchery because it probably retains the genetic resources of steelhead populations that once occurred above the Grand Coulee Dam. This species does not include the Skamania Hatchery stock because of its non-native genetic heritage.

Listing status

Upper Columbia River steelhead were listed as endangered under the ESA in 1997. Critical habitat designation for this ESU was vacated on May 7, 2002.

Population status and trends

Returns of Upper Columbia River natural-origin steelhead to Priest Rapids dam have declined from a 4-year average of 2,900 (beginning in 1986-1987) to 900 (present) although escapements appear to have stabilized at a range of 800-900, over the past six years. Hatchery populations of Upper Columbia River steelhead are included in the species and are also listed as endangered. The hatchery component is relatively abundant and usually, exceeds hatchery supplementation program needs by a substantial margin.

The naturally spawning population of Upper Columbia River steelhead has been augmented for a number of years by stray hatchery fish that have spawned naturally. Replacement ratios for naturally spawning fish (natural-origin and hatchery strays) are quite low, on the order of 0.3. This very low return rate suggests that the productivity of the river basin is so low hatchery strays have been supporting the population.

Impacts of human activity on this species

When this species was listed, the Biological Review Team that reviewed the status of this species concluded that Upper Columbia steelhead were presently in danger of extinction. While total abundance of populations within this species has been relatively stable or increasing, this appears to be occurring only because of major hatchery supplementation programs. Estimates of the proportion of hatchery fish in spawning escapement are 65% (Wenatchee River) and 81% (Methow and Okanogan Rivers). Their major concern for this species was the clear failure of natural stocks to replace themselves. They were also concerned about problems of genetic homogenization due to hatchery supplementation within the species and about the apparent high harvest rates on steelhead smolts in rainbow trout fisheries and the degradation of freshwater habitats within the region, especially the effects of grazing, irrigation diversions, and hydroelectric dams.

Middle Columbia River Steelhead

Species description and distribution

Middle Columbia steelhead occupy the Columbia River Basin from Mosier Creek, Oregon, upstream to the Yakima River, Washington, inclusive (61 FR 41541; August 9, 1996). Steelhead from the Snake River Basin (described elsewhere) are excluded. This species includes the only populations of inland winter steelhead in the United States, in the Klickitat River and Fifteenmile Creek (Busby *et al*. 1996). Two hatchery populations are considered part of this species, the Deschutes River stock (ODFW stock 66) and the Umatilla River stock (ODFW stock number 91); listing for neither of these stocks was considered warranted.

Most Middle Columbia River steelhead smolt at 2 years and spend 1 to 2 years in salt water (i.e., 1-ocean and 2-ocean fish, respectively) prior to re-entering fresh water, where they may remain up to a year prior to spawning (Howell *et al*. 1985). Within this species, the Klickitat River is unusual in that it produces both summer and winter steelhead, and the summer steelhead are dominated by 2-ocean steelhead, whereas most other rivers in this region produce about equal numbers of both 1-and 2-ocean steelhead.

Listing status

Middle Columbia River steelhead were listed as endangered under the ESA in 1999. The critical habitat designation for this ESU was vacated on May 7, 2002.

Population status and trends

Populations of Middle Columbia River steelhead in the Yakima, Umatilla and Deschutes River basins appear to be increasing. Part of the reason for listing this species as threatened were low returns to the Yakima River, low estimates of winter steelhead abundance in Klickitat River and Fifteenmile Creek, and an overall decline of naturally-producing stocks.

Impacts of human activity on this species

Middle Columbia River steelhead occupy the intermontane region which includes some of the driest areas of the Pacific Northwest, generally receiving less than 40 cm of rainfall annually. Vegetation is of the shrub-steppe province, reflecting the dry climate and harsh temperature extremes. Because of this habitat, occupied by the species, factors contributing to the decline include agricultural practices, especially grazing, and water diversions and withdrawals. In addition, hydropower development has impacted the species by preventing these steelhead from migrating to habitat above dams, and by killing them in large numbers when they try to migrate through the Columbia River hydroelectric system.

Lower Columbia River Steelhead

Species description and distribution

Lower Columbia River steelhead include naturally-produced steelhead returning to Columbia River tributaries on the Washington side between the Cowlitz and Wind rivers in Washington and on the Oregon side between the Willamette and Hood rivers, inclusive. In the Willamette River, the upstream boundary of this species is at Willamette Falls. This species includes both winter and summer steelhead. Two hatchery populations are included in this species, the Cowlitz Trout Hatchery winter-run stock and the Clackamas River stock (ODFW stock 122) but neither was listed as threatened.

Listing status

Lower Columbia River steelhead were listed as threatened under the ESA in 1998. The critical habitat designation for this ESU was vacated on May 7, 2002.

Population status and trends

There are no historical estimates of this species' abundance. Because of their limited distribution in upper tributaries and urbanization in the lower tributaries (e.g., the lower Willamette, Clackamas, and Sandy Rivers run through Portland or its suburbs), habitat degradation appears to have threatened summer steelhead more than winter steelhead. Steelhead populations in the lower Willamette, Clackamas, and Sandy Rivers appear stable or slightly increasing although sampling error limits the reliability of this trend. Total annual run size data are only available for the Clackamas River (1,300 winter steelhead, 70% hatchery; 3,500 wild summer steelhead).

Upper Willamette River Steelhead

Species description and distribution

Upper Willamette River steelhead occupy the Willamette River and its tributaries upstream of Willamette Falls. This is a late-migrating winter group that enters fresh water in March and April (Howell *et al*. 1985). Only the late run was included is the listing of this species, which is the largest remaining population in the Santiam River system.

Listing status

Upper Willamette River steelhead were listed as threatened under the ESA in 1999. The critical habitat designation for this ESU was vacated on May 7, 2002.

Population status and trends

No estimates of abundance prior to the 1960s are available for this species. Recent run size can be estimated from redd counts, dam counts, and counts at Willamette Falls (late stock). Recent total-basin run size estimates exhibit general declines for winter steelhead. The majority of winter steelhead populations in this basin may not be self-sustaining.

Impacts of human activity on this species

A major threat to Willamette River steelhead results from artificial production practices. Fishways built at Willamette Falls in 1885 have allowed Skamania-stock summer steelhead and early-migrating winter steelhead of Big Creek stock to enter the range of Upper Willamette River steelhead. The population of summer steelhead is almost entirely maintained by hatchery salmon, although natural-origin, Big Creek-stock winter steelhead occur in the basin (Howell *et al*. 1985). In recent years, releases of winter steelhead are primarily of native stock from the Santiam River system.

Snake River Basin Steelhead

Species description and distribution

Snake River basin steelhead are an inland species that occupy the Snake River basin of southeast Washington, northeast Oregon, and Idaho. The historic spawning range of this species included the Salmon, Pahsimeroi, Lemhi, Selway, Clearwater, Wallowa, Grande Ronde, Imnaha, and Tucannon Rivers.

Life history information

Snake River Basin steelhead, like most inland steelhead, are “summer-run” which means they enter freshwater nine or ten months before spawning. Snake River Basin steelhead enter fresh water from June to October and spawn in the following spring from March to May. The two components, A-run and B-run, are distinguished by their size, the timing of their respective adult migrations, and ocean-age. Because of these timing differences, the A-run component of the Snake River Basin steelhead is most affected by the winter, spring, and summer season fisheries in the Columbia River.

Listing status

Snake River steelhead were listed as threatened under the ESA in 1997. The critical habitat designation for this ESU was vacated on May 7, 2002.

Population status and trends

No estimates of historical (pre-1960s) abundance specific to Snake River steelhead are available. An estimated 80% of the total Columbia River Basin steelhead that run above Bonneville Dam (summer and winter steelhead combined) are hatchery fish. Total recent 5-year average escapement above Lower Granite Dam was approximately 71,000, with a natural component of 9,400 (7,000 A-run and 2,400 B-run).

Impacts of human activity on this species

When this species was listed, the Biological Review Team that reviewed the status of this species concluded that Snake River Basin steelhead were not presently in danger of extinction, but were likely to become endangered in the foreseeable future (although some members of the team concluded that there was little likelihood that this ESU will become endangered).

Although the total (hatchery + natural) run size has increased since the mid-1970s, Snake River Basin steelhead recently experienced severe declines in natural run sizes. The majority of natural stocks of this species have been declining. Parr densities in natural production areas have been substantially below estimated capacity in recent years. Downward trends and low parr densities indicate a particularly severe problem for B-run steelhead, whose loss would substantially reduce life history diversity of Snake River basin steelhead.

Northern California Steelhead

Species description and distribution

The Northern California steelhead ESU includes steelhead in California coastal river basins from Redwood Creek south to the Gualala River, inclusive. Major river basins containing spawning and rearing habitat for this ESU comprise approximately 6,672 square miles in California.

Listing status

Northern California steelhead were listed as threatened under the ESA in 2000. Critical habitat has not been designated for this ESU.

Population status and trends

Population abundances are very low relative to historical estimates. While no overall recent abundance estimates are available for the ESU, counts at Cape Horn Dam have declined from 4400 adults in the 1930's to an average of 30 wild adults in 1996.

Central California Coast Steelhead

Species description and distribution

The Central California Coast steelhead ESU includes steelhead in river basins from the Russian River to Soquel Creek, Santa Cruz County (inclusive) and the drainages of San Francisco and San Pablo bays; excluded is the Sacramento-San Joaquin River Basin of the Central Valley of California.

Listing status

Northern California steelhead were listed as threatened under the ESA in 2000. The critical habitat designation for this ESU was vacated on May 7, 2002.

Population status and trends

Abundance in the Russian and San Lorenzo Rivers, the areas with the two largest steelhead stocks is approximately 15% of what it was historically. There are no recent estimates of abundance for this ESU.

South-Central California Coast Steelhead

Species description and distribution

The South-Central California steelhead ESU includes all naturally spawned populations of steelhead (and their progeny) in streams from the Pajaro River (inclusive) to, but not including the Santa Maria River, California.

Listing status

South-Central California Coast steelhead were listed as threatened under the ESA in 1997. The critical habitat designation for this ESU was vacated on May 7, 2002.

Population status and trends

While we have no recent estimates of total run size for this ESU, recent run-size estimates are available for five streams. The total of these estimates is less than 500, compared with a total of 4,750 for the same streams in 1965, which indicates a substantial decline for the entire ESU from 1965 levels.

Southern California Steelhead

Species description and distribution

This coastal steelhead ESU occupies rivers from the Santa Maria River to the southern extent of the species range.

Listing status

Southern California steelhead were listed as endangered under the ESA in 1997. The critical habitat designation for this ESU was vacated on May 7, 2002.

Population status and trends

Currently, there are no run sizes in this ESU with greater than 200 adults. The populations are extremely depressed and steelhead have been extirpated from many streams in the ESU.

California Central Valley Steelhead

Species description and distribution

This steelhead ESU occupies the Sacramento and San Joaquin Rivers and their tributaries.

Listing status

California Central valley steelhead were listed as threatened under the ESA in 1998. The critical habitat designation for this ESU was vacated on May 7, 2002.

Population status and trends

Historically, steelhead occurred naturally throughout the Sacramento and San Joaquin River Basins; however, stocks have been extirpated from large areas of the Sacramento River Basin and possibly from the entire San Joaquin River Basin. Recent abundance estimates are less than 10,000 individuals in the basin.

Ambient Noise in the Ocean Environment

Ambient noise is the typical or persistent environmental background noise that is present. When it is measured using listening devices, it does not include "self noise" generated by the listening devices or the vessel on which they are mounted or any sound source that can be identified (such as a passing ship). Ambient noise is directional both horizontally and vertically, meaning that it does not come at equal sound levels from all directions.

Measurements of ambient noise have been made over frequency ranges from below 1 Hz to 100 kHz. Ambient noise levels and sources vary both in location and season. Ambient noise levels are higher in the northern hemisphere, where sources of anthropogenic sounds are more pervasive. However, even in relatively quiet regions in the southern hemisphere, ambient noise levels will commonly vary by 20 dB and will vary by 30 dB with lower frequency because of biological sources and sea surface noise (Cato and McCauley 2001). There are numerous ambient noise sources that are comparable in frequency to SURTASS LFA sonar. Distant shipping noise has been reported by Urick (1983) to be between 20 and 300 Hz, and by Richardson *et al*. (1995b) to be from 50 to 500 Hz. Biological noise can also be a major contributor of noise in the ocean. Several species of baleen whales, toothed whales, and seals are known to produce underwater sounds between 100 and 500 Hz (Table 2).

Wind and Waves

Wind and waves are common and interrelated sources of ambient noise in all of the world's oceans. All other factors being equal, ambient noise levels tend to increase with increasing wind speed and wave height (Richardson *et al*. 1995). Noise generated by surface wave activity and biological sounds is the primary contributor over the frequency range from 300 Hz to 5 kHz. The wind-generated noise level decreases smoothly with increasing acoustic frequency (i.e., there are no spikes at any given frequency).

Precipitation

At some frequencies, rain and hail will increase ambient noise levels. Significant noise is produced by rainsqualls over a range of frequencies from 500 Hz to 15 kHz. Large storms with heavy precipitation can generate noise at frequencies as low as 100 Hz and significantly affect ambient noise levels at a considerable distance from a storm's center. Lightning strikes associated with storms are loud, explosive events that deliver an average of 100 kilojoules per meter (kJ/m) of energy (Considine 1995). Hill (1985) estimated the source level for cloud-to-water pulse to be 260.5 dB. It has been estimated that over the earth's oceans the frequency of lightning averages about 10 flashes per second, or 314 million strikes per year (Kraght 1995).

Seismic Phenomena

In the Pacific Ocean, about 10,000 natural, seismic phenomena like earthquakes, underwater volcanic eruptions, and landslides occur each year (Fox *et al.* 2001). These phenomena produce sounds with source levels exceeding 210 dB.

Table 2: Natural sources of low frequency sounds in marine environments

Source	Broadband Levels	References
Lightning Strike on Water Surface		Hill 1985
Seafloor Volcanic Eruption		Dietz and Sheehy 1954; Northrop 1974
Sperm Whale		Levenson 1974; Watkins 1980a
Fin Whale	155 - 186 dB	Watkins 1981; Edds 1988; Watkins, *et al*. 1987; Cummings & Thompson 1994
Humpback Whale	144 - 174 dB	Thompson, *et al*. 1979; Payne and Payne 1985; Frankel 1994
Bowhead Whale	128 - 189 dB	Ljungblad, *et al*. 1982a; Cummings and Holliday 1987; Würsig and Clark 1993
Blue Whale	155 - 188 dB	Aroyan *et al*., 2000; Cummings and Thompson 1971 and 1994; Edds 1982; Stafford *et al*. 1994
Southern Right Whale		Cummings, *et al*. 1972; Clark 1982, 1983
Gray Whale	142 - 185 dB	Cummings, *et al*. 1968; Fish, *et al*. 1974; Swartz and Cummings 1978

Biological Noises

Biological noises are sounds created by animals in the sea and may contribute significantly to ambient noise in many areas of the oceans (Curtis *et al.* 1999). Because of the habits, distribution, and acoustic characteristics of these sound producers, certain areas of the oceans are louder than others. Only three groups of marine animals are known to make sounds: crustaceans (such as snapping shrimp), true fish, and marine mammals (Urick 1983).

The most widespread, broadband noises from animal sources (in shallow water) are those produced by croakers (representative of a variety of fish classified as drumfish) (100 Hz to 10 kHz) and snapping shrimp (500 Hz to 20 kHz). Sound-producing fishes and crustaceans are restricted almost entirely to bays, reefs, and other coastal waters, although there are some pelagic, sound-producing fish. In oceanic waters, whales and other marine mammals are principal contributors to biological noise. For example, dolphins produce whistles associated with certain behaviors, and the baleen whales are noted for their low frequency vocalizations.

Human Activity

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 transportation, dredging, construction; oil, gas, and mineral exploration in offshore areas; geophysical (seismic) surveys; sonars; explosions; and ocean research activities (Richardson *et al*. 1995). Table 3 shows the source levels for selected sources of anthropogenic low frequency underwater noise.

Surface shipping is the most widespread source of anthropogenic, low frequency (0 to 1,000 Hz) noise in the oceans (Simmonds and Hutchinson 1996). 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 at sea at any one time (U.S. Navy 2001). The radiated noise spectrum of merchant ships ranges from 20 to 500 Hz and peaks at approximately 60 Hz. Ross (1976) has estimated that between 1950 and 1975 shipping had caused a rise in ambient ocean noise levels of 10 dB. 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.

Table 3. Summary and comparison of source levels for selected sources of anthropogenic low frequency underwater noise

Sound Source (Transient)		Source Level in dB
Seismic Survey - Air gun array (32 guns) (Impulsive - Peak)		259[1] Broadband
Explosions (Impulsive)		
0.5 kg (1.1 lb) TNT		267[1] Broadband
2 kg (4.4 lb) TNT	Peak	271[1] Broadband
20 kg (44 lb) TNT	Peak	279[1] Broadband
4,536 kg (10,000 lb) TNT	Peak	>294[2] Broadband
Ocean Acoustics Studies		
Heard Island Test		220[1] Spectrum Level
ATOC		195[1] Spectrum Level
Vessels Underway		
Tug and Barge (18 km/hour)		171[1] Broadband
Supply Ship (*Kigoriak*)		181[1] Broadband
Large Tanker		186[1] Broadband
Icebreaking		193[1] Broadband
Notes: All dB re 1 μPa at 1 m. Sources: 1. Richardson *et al*. 1995b. 2. Urick 1983.		

Michel *et al.* (2001) suggested an association between long-term exposure to low frequency sounds from shipping and an increased incidence of marine mammal mortalities caused by collisions with shipping. At lower frequencies, the dominant source of this noise is the cumulative effect of ships that are too far away to be heard individually, but because of their great number, contribute substantially to the average noise background.

Seismic survey airguns have source levels reaching and exceeding 250 dB (Richardson *et al.* 1995b), with a "shot" every 15 seconds, or 240 shots per hour, 24 hours per day. Each airgun shot of a few milliseconds contains less acoustic energy than a single, 60-second SURTASS

LFA sonar ping. Seismic survey airguns operate seven days a week. For example, a seismic survey vessel normally works for at least two weeks straight, producing almost 81,000 shots. In the Gulf of Mexico alone, there are typically three such survey vessels operating on any given day and over 100 seismic surveys are conducted there each year. Underwater recordings evaluated by Fox *et al.* (2001) from the central North Atlantic Ocean since 1999 were dominated by noise from seismic airguns working off Canada, Brazil, and West Africa.

Deep Water 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. 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 ambient noise frequency spectrum and level 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.

Shallow Water Ambient Noise

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.

EFFECTS OF THE ACTION

This biological opinion assesses the effects of the Navy's proposed employment of the SURTASS LFA sonar on threatened and endangered species. This includes the Navy's proposal to incorporate mitigative measures to avoid potential, adverse effects of SURTASS LFA sonar transmissions on threatened or endangered species. In the *Description of the Action* section of this Opinion, NMFS provided an overview of the Navy's proposed program and NMFS' proposed permit that would authorize incidental taking of cetaceans and cetaceans by the SURTASS LFA sonar for the purposes of the MMPA. In the *Status of the Species* section of this Opinion, NMFS provided an overview of the species that may be adversely affected by the SURTASS LFA sonar.

In this section of a biological opinion, NMFS assesses the probable direct and indirect effects of the SURTASS LFA sonar operations and of interrelated, and interdependent actions on threatened and endangered species and designated critical habitat. The purpose of this assessment is to determine if it is reasonable to expect that the Navy's SURTASS LFA sonar will have direct or indirect effects on threatened and endangered species that appreciably reduce their likelihood of surviving and recovering in the wild [which is the jeopardy standard established by 50 CFR 402.02]. This assessment also determines if it is reasonable to expect the Navy's SURTASS LFA sonar will result in the destruction or adverse modification of critical habitat.

NMFS generally approaches these analyses by first evaluating the available evidence to identify the direct and indirect physical, chemical, and biotic effects of a proposed action on individual members of listed species or aspects of a species' environment. Once these effects have been identified, NMFS then evaluates the available evidence to identify a species' probable responses (including behavioral responses) to those effects to determine if those effects could reasonably be expected to reduce a species' reproduction, numbers, or distribution (for example, by changing birth, death, immigration, or emigration rates; increasing the age at which individuals reach sexual maturity; decreasing the age at which individuals stop reproducing; among others). NMFS then uses the evidence available to determine if these reductions, if there are any, could reasonably be expected to reduce a species' likelihood of surviving and recovering in the wild. If NMFS concludes that an action could reasonably be expected to reduce a species' likelihood of surviving and recovering in the wild, NMFS' final task is determining whether that reduction is likely to be "appreciable."

Approach to the Assessment

NMFS identified four aspects of the SURTASS LFA sonar system that represent potential hazards to listed species or their critical habitat: (1) the ship associated with the SURTASS LFA system; (2) the surface-towed array sonar system (SURTASS); (3) the low-frequency active (LFA) sonar; and (4) the high-frequency marine mammal monitoring (HF/M3) system. The ship (Element 1) represents a potential hazard to listed species and their critical habitat because of potential ship strikes and the generation of engine and propeller noise. The SURTASS array (Element 2) does not represent a potential hazard (that is, it is not likely to affect listed species or their designated critical habitat) because it is a passive system and is not likely to strike or entangle a whale or sea turtle (it is a plastic tube that is towed behind vessels); therefore, we will not consider this aspect of the proposed action further. The LFA sonar and HF/M3 sonar (Elements 3 and 4) pose potential risks to listed species and their critical habitat by generating sounds that could potentially injure listed species or result in behavioral changes that alter their reproductive success or survival.

NMFS examined these three aspects of the SURTASS LFA sonar system to (1) identify possible interactions with listed species or designated critical habitat (that is, their exposure to the LFA sonar source at different received levels); (2) determine the susceptibility of any listed species that was exposed to one or more aspect of the SURTASS LFA system (by susceptibility, we mean having a constitution or temperament that is open, subject, or unresistant to an agency, influence, intervention, or stimulus); and (3) identify the probable response of a listed species or critical habitat that was exposed and sensitive to one or more aspect of the sonar system. Because we did not have specific information on the probable responses of every threatened or endangered species to anthropogenic, low frequency sounds, we completed some of the analyses for this step by focusing on species groups of toothed whales and baleen whales. Specifically, we evaluated the available evidence to determine if the SURTASS LFA sonar system would be expected to physically injure, harm, or harass listed species or destroy or adversely modify critical habitat that has been designated for them.

As indicated in the *Status of the Species* section of this Opinion, NMFS assessed the effects of SURTASS LFA on threatened and endangered species and critical habitat using a generalized assessment model patterned after the U.S. Environmental Protection Agency's *Guidelines for Risk Assessment* (U.S. EPA 1998) and models toxicologists and epidemiologists use to assess risks posed by terrestrial, aquatic, and atmospheric pollutants (Kapustka *et al*. 1996, Landis *et al*. 1994, Landis *et al*. 1997, Lipton *et al*. 1993, McCarty and Power 1997, Newman *et al*. 2000, Norton *et al*. 1992, Taub 1997, U.S. EPA 1998, Wentsell 1994). In effect, this assessment evaluates SURTASS LFA sonar as a potential "pollutant" in the ocean environment. The first step of our analysis evaluates the available evidence to determine the likelihood of listed species or critical habitat being exposed to sound pressure levels associated with SURTASS LFA sonar, which includes estimating the intensity, duration, and frequency of exposure (for other examples of exposure assessments, see Wu and Schaum 2000). Our analysis assumed that SURTASS LFA sonar poses no risk to listed species or critical habitat if they are not exposed to sound pressure levels from the SURTASS LFA sound source (we recognize that LFA sonar could have indirect, adverse effects on listed species or critical habitat by disrupting marine food chains, a species' predators, or a species' competitors; however, we could not identify situations where this concern would be applicable to species under NMFS' jurisdiction). These analyses also assume that the potential effects of SURTASS LFA sonar on a species would be a function of the intensity (measured in both sound pressure level in decibels and frequency), duration, and frequency of exposure to the SURTASS LFA sound source.

For species or critical habitat that were likely to be exposed to LFA sonar at received levels that could elicit responses, the second step of our analyses evaluated the available evidence to identify probable responses of listed species to SURTASS LFA sonar or, alternatively, the potential effects of differing received levels on listed species based on the species' susceptibility to those received levels (measured in decibels at 1 μPa) at the frequencies associated with SURTASS LFA sonar. For example, this section examines "stressor-response" relationships to characterize the potential ecological effects of SURTASS LFA sonar on listed species and designated critical habitat. This step of our assessment relied on the Navy's acoustic models and risk continuum analyses (see U.S. Navy 2001a, 2001b, 2001c), which estimate the risk posed by SURTASS LFA sonar by correlating the risk of biologically significant behavior to received levels (single ping equivalents in decibels) using probability distribution functions. The results of the Navy's analyses are presented as continuous functions that are analogous to "stressor-response" curves (called "dose-response" curves in epidemiology and toxicology): at one end of these curves, low received levels ("low dose") would not be expected to elicit a negative response in a species; at the other end of these curves, high received levels ("high dose") would be expected to elicit much more serious, negative responses.

For species or critical habitat that were likely to be exposed to LFA sonar and, if exposed, were susceptible to the sonar, the final step of our assessment estimates the probable risks posed by SURTASS LFA sonar by integrating and synthesizing the information on exposure and responses to reach conclusions about potential effects of SURTASS LFA sonar on listed species and critical habitat. Although several authors have proposed several measures of the effect of low frequency sounds on marine mammals (for examples, see Southern *et al.* 2001 and Bain *et al.* 2001), most of those procedures are still in the developmental stages or the available data precluded their use in this assessment. As a result, we used more traditional measures of effects: potential injury, resonance effects, stranding, masking, and biologically-important behavioral responses. Specifically, we evaluated the available evidence to determine if LFA sonar was likely to (1) physically injure marine species in a way that would have acute or chronic effects or (2) elicit behavioral responses that would have longer-term, chronic effects on the viability of populations of a species (as discussed previously, we initially screened for indirect effects resulting from the disruption of food chains or interactions between species, but our searches produced no evidence of these phenomena). Although this section of our assessment included concerns for effects on individual animals, our assessment focused on the probable effects of SURTASS LFA sonar on populations and, through populations, listed species.

Because it was technically, logistically, and financially impossible to study the potential effects of LFA sonar on all marine animals, the Navy's Low Frequency Sound Scientific Research Program (Scientific Research Program) focused on four mysticete species (blue, fin, gray, and humpback whales). These animals were selected because: (1) based on evaluations of earlier studies, these species, of all marine mammals, were considered most likely to have the best hearing in the SURTASS LFA sonar frequency band, (2) all of these species are protected under the ESA, and (3) there was prior evidence to suggest that these species avoided low frequency sounds. The responses of these species to low frequency sound signals during the Scientific Research Program were used as indicators of the responses of other species that were presumed to be less sensitive to SURTASS LFA sonar signals. The rationale for using representative species to study the potential effects of low frequency sound on marine animals emerged from an extensive review in several workshops by a broad group of interested parties: academic scientists, federal regulators, and representatives of environmental and animal welfare groups. The outcome of these discussions concluded that baleen whales (mysticetes) would be the focus of the three phases of the Scientific Research Program and indicator species for other marine animals in the analysis of underwater acoustic impacts. Because of this, effects analyses based on these species may overestimate the potential effects on LFA sonar on other, less-sensitive marine mammals.

NMFS defines harm to 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). The ESA does not define harassment nor has NMFS defined this term, pursuant to the ESA, through regulation. However, the MMPA, as amended, defines harassment as any act of pursuit, torment, or annoyance which has the potential to injure a marine mammal or marine mammal stock in the wild or has the potential to disturb a marine mammal or marine mammal stock in the wild by causing disruption of behavioral patterns, including, but not limited to, migration, breathing, nursing, breeding, feeding, or sheltering [16 U.S.C. 1362(18)(A)]. The latter portion of this definition (that is "...causing disruption of behavioral patterns including... migration, breathing, nursing, breeding, feeding, or sheltering") is almost identical to the U.S. Fish and Wildlife Service's regulatory definition of "harass"[3] under the ESA.

For this biological opinion, we will define harassment as a disturbance resulting from a human action that disrupts one or more behavioral patterns that are essential to an individual animals' life history or to the animal's contribution to a population, or both. We are particularly concerned about injuries that may manifest themselves as animals that fail to feed successfully, breed successfully (which can result from feeding failure), or complete life history patterns normal to their species because of changed behavior. In the latter two of these examples, the effects on individual animals could disadvantage a population because the animals breeding success would have been reduced (assuming also that the number of individuals affected would affect the population).

Uncertainty

To conduct this assessment of the effects of SURTASS LFA sonar on threatened and endangered species, NMFS examined an extensive amount of evidence from a variety of sources. Many investigators have studied potential responses of marine mammals and other marine organisms to human-generated sounds in marine environments (for example, Bowles *et al*. 1994; Croll *et al*. 1999, 2001; Frankel and Clark 1998; McCauley and Cato 2001; Norris 1994; Tyack 2000; Whitlow *et al*. 1997). We supplemented these studies with literature using First Search and Biosis searches on the biology and ecology of marine mammals, their responses to low frequency sound, and the general effects of sounds on wildlife.

Despite these studies, this assessment involved a large amount of uncertainty. We lacked information on the probable responses of different species of marine mammals, including the variability of those responses, to various human-generated sounds in the marine environment. We also lacked information on the consequences of those responses to the population ecology of different marine mammals; that is, we had information suggesting statistically-significant changes in the length of a humpback whale's song in the presence of human-generated sound, but we did not have information that allowed us to determine if those statistically-significant changes were also significant biologically; that is, we did not have information that related those changes to acute or chronic changes in the population's survival, reproductive success, or longevity. We still need more information on the basic hearing capabilities of marine mammals, on how marine mammals use natural sound to communicate and its importance to their normal behavioral routines, on whether low-frequency sounds affect marine mammal behavior and physiology (including the non-auditory physiology), and on sound pressure levels that produce temporary and permanent hearing loss in marine mammals (see NRC 2000 for further discussion of these unknowns).

Finally, we lacked information on how marine mammals interpret sounds generally, including human-generated sounds, and how sounds affect their cognitive processes and their behavior. On

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[3] An intentional or negligent act or omission which creates the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering (50 CFR 17.4)

a basic level, we do not know – and, perhaps, cannot know – how marine mammals interpret various sounds in the ocean environment and the relationship between those interpretations and marine mammal behavior. For example, we do not know if or when marine mammals would classify a sound as a pollutant in their environment (noise), although this information is relevant to an analysis of the effects of SURTASS LFA sonar on threatened and endangered species. Despite its relevance, this information is not available for this biological opinion and, for obvious reasons, may never become available. Therefore, while we recognize the limitations of the available data, we have drawn conclusions from the best available scientific information on the physics of low frequency sounds in the ocean environment and current knowledge of how marine mammals respond to low frequency sounds generated by humans.

The primary sources of information on the effects of low frequency sound on marine mammals were reviews conducted by the National Research Council (NRC 1994, 1996, 2000) and Richardson *et al*. (1995) on marine mammals and noise, the Navy's Low Frequency Sound Scientific Research Program (which was developed to address questions associated with SURTASS LFA sonar), Marine Mammal Research Program (which was developed to address questions associated with the Advanced Research Projects Agency's Acoustic Thermometry of Ocean Climate project, which also uses low frequency sound), several models the Navy developed for its Environmental Impact Statement on SURTASS LFA sonar, and numerous scientific papers (Croll *et al*. 1999 and 2001; Frankel and Clark 1998; Richardson *et al*. 1995; Tyack 2000; Whitlow *et al*. 1997).

NMFS also had the results of the Navy's acoustic models and risk continuum analyses available for this assessment (see U.S. Navy 2001a, 2001b, 2001c). The Navy's analyses estimated the risk posed by SURTASS LFA sonar by treating the risk of biologically significant behavior to received levels (single ping equivalents in decibels) using probability distribution functions. The results of these analyses appear as continuous functions that are analogous to stressor-response curves (called "dose-response" curves in epidemiology and toxicology): at one end of these curves, low received levels ("low dose") would not be expected to elicit a response in the species; at the other end of these curves, high received levels ("high dose") would be expected to elicit a much more serious responses. These analyses are accepted as the best practice in disciplines ranging from epidemiology, toxicology, and pharmacology (Hill 1965, Fox 1991, U.S. EPA 1996, U.S. EPA 1998); these analyses also made it possible to approach this assessment using approaches that were developed in those other disciplines.

Assumptions to Overcome Limitations in the Information Available

To conduct our effects analyses in the face of the limitations we discussed in the preceding section, we made one major assumption: we assumed that the response of a species' group would represent the response of the individual species in the group. For example, virtually no information was available on the responses of northern right whales (*E. glacialis* and *E. japonica*) to anthropogenic low frequency sounds, generally, or SURTASS LFA sonar in particular, so we assumed that the responses of these whales to low frequency sounds would be similar to the responses of baleen whales generally (all of these species were grouped as Mysticetes). Similarly, we grouped the various species of toothed whales (Odontocetes) for the purposes of this assessment (see Ketten 1997 for support for this assumption).

Effects of the Vessels Associated with SURTASS LFA Sonar

Operation of the vessels supporting SURTASS LFA could pose potential risk to listed species and their critical habitat through discharges, ship strikes, and the generation of engine and propeller noise. Operation of the vessels supporting SURTASS LFA will result only in discharges incidental to normal operations of a vessel. The International Convention for the Prevention of Pollution from Ships (MARPOL 73/78) prohibits certain discharges of oil, garbage, and other substances from vessels. The Convention is implemented by the Act to Prevent Pollution from Ships (APPS; 33 U.S.C. 1901 et seq.), which establishes requirements for the operation of U.S. Navy vessels. The vessels supporting the SURTASS LFA sonar system will operate in compliance with these requirements. The sonar system itself will not result in the discharge of any pollutants regulated under APPS.

In addition, vessel speeds are approximately 3 knots (3.5 miles per hour) during SURTASS LFA sonar operations and about 10 knots (11.6 miles per hour) during transit. Because of these slow speeds, the vessels should produce far less engine and propeller noise than commercial and recreational vessels.

Furthermore, during SURTASS LFA sonar operations the tripartite monitoring mitigation would be expected to virtually eliminate the risk of ship strikes. Therefore, ship strikes are unlikely. Therefore, the ships pose negligible threats to listed species or critical habitats.

Effects of SURTASS LFA Sonar

This section of the Opinion begins with an assessment of the potential exposure of listed species and critical habitat to sound pressure levels associated with SURTASS LFA sonar, including estimates of the intensity, duration, and frequency of any exposure. For listed species or critical habitat that are likely to be exposed to LFA sonar, the second step of our approach assesses the sensitivities of listed species to SURTASS LFA sonar or, alternatively, the potential effects of differing levels of low-frequency sound on listed species ("stressor-response" relationships). For species or critical habitat that are likely to be exposed to LFA sonar and, if exposed, were sensitive to the sonar, the third step of our approach estimated the probable risks posed by SURTASS LFA sonar based on exposure estimates and sensitivities to reach conclusions about the potential effects of SURTASS LFA sonar on species and critical habitat (also known as "risk estimation").

Exposure Analysis

This exposure analysis evaluates the available evidence to estimate the potential exposure of listed species and critical habitat to signals generated by SURTASS LFA sonar. This analysis includes an evaluation of the intensity, duration, and frequency of a species' exposure to different sound pressure levels associated with SURTASS LFA sonar. The analysis assumed that exposure to SURTASS LFA sonar, regardless of the sound pressure level, is a pre-requisite for listed species or critical habitat: listed species or critical habitat to be adversely affected by the sonar.[4]

As discussed in the *Description of the Action* section of this Opinion, the LFA sonar system consists of up to 18 low-frequency acoustic-transmitting projectors that are suspended from a cable beneath a ship. The source level of an individual projector in the LFA sonar array is approximately 215 dB, and the sound field of the array can never have a sound pressure level higher than that of an individual projector. The typical LFA sonar signal is not a constant tone, but is a transmission of various signal types that vary in frequency and duration (including continuous wave and frequency-modulated signals). The Navy refers to a complete sequence of sound transmissions as a "ping" which can range from between 6 and 100 seconds, with no more than 10 seconds at any single frequency. The time between pings will typically range from 6 to 15 minutes. The Navy can control the average duty cycle (the ratio of sound "on" time to total time) for the system but the duty cannot exceed 20 percent; the Navy anticipates a typical duty cycle between 10 and 15 percent.

Before we assess the potential exposure of marine mammals to the LFA sonar signal, we first summarize information on how that signal would propagate through a marine ecosystem. For more complete analyses of sound propagation in marine environments, readers should refer to Richardson *et al*. (1995) and Appendix B of the Navy's EIS on SURTASS LFA (Navy 2001).

Propagation of the SURTASS LFA Sonar Signal

Sound is a wave of pressure variations propagating through a medium (for SURTASS LFA, the medium is marine water). 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 national standard reference pressure for underwater sound is 1 microPascal (μPa), for airborne sound, the standard reference pressure is 20 μPa (Richardson *et al*., 1995).

In addition, 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 microPascal (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). Humans perceive a 10 dB increase in noise as a doubling of sound level, or a 10 dB decrease in noise as a halving of sound level. The term "sound pressure level" implies a decibel measure and a reference pressure that is used as the denominator of the ratio.

Sound propagation in water is influenced by various physical characteristics, including water temperature, depth, salinity, and surface and bottom properties that cause refraction, reflection,

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[4] As discussed previously, we recognize that LFA sonar could have indirect, adverse effects on species or critical habitat by disrupting marine food chains, a species predators, or a species competitors; however, we could not identify a situation where this concern would be applicable to species under NMFS' jurisdiction.

absorption, and scattering of sound waves. Oceans are not homogeneous and the contribution of each of these individual factors is extremely complex and interrelated. The physical characteristics that determine the sound's speed through the water will change with depth, season, geographic location, and with time of day (as a result, in actual LFA sonar operations, crews on the SURTASS LFA platform will measure oceanic conditions, such as sea water temperature versus depth, every 12 hours to calibrate models that will determine the path the low-frequency signal will take as it travels through the ocean and how strong the sound signal will be at given range along a particular transmission path).

Sound tends to follow many paths through the ocean, so that a listener would hear multiple, delayed copies of transmitted signals (Navy 2001, Richardson *et al*. 1995). 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. These are called acoustic rays. 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 are curved.

Sound speed in seawater is about 1,500 m/s (5,000 ft/s) and 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.

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. As sound travels through the ocean, the intensity associated with the wavefront diminishes, or attenuates. This decrease in intensity is referred to as propagation loss, also commonly called transmission loss.

A major component of transmission loss is spreading loss (Navy 2001, Richardson *et al*. 1995). From a point source in a uniform medium, sound spreads outward as spherical waves; transmission losses due to spherical spreading, in dB, is represented, in simplified form, as 20 $\log_{10}$ R (= radius from source). Using this equation, a 215 dB signal from a SURTASS LFA projector would be expected to attenuate by about 60 dB one kilometer from the source and by about 66 dB two kilometers from the source. Cylindrical spreading sometimes occurs when the medium is not homogenous. In shallow water, sound will reflect from the ocean's surface and the ocean bottom. With cylindrical spreading, sound levels diminish by 3 dB when distance doubles and by 10 dB when distance increases 10-fold (Richardson *et al*. 1995).

Near an ocean's surface, mixing results in a fairly constant temperature and salinity. In surface areas of this mixed layer (roughly the uppermost 150 feet), the sound field will be dominated by sound generated by wave action, rain, and other surface activity; which would mask sounds generated by SURTASS LFA sonar. Below the surface area of this mixed layer, depth (pressure) dominates the sound speed profile and the sound's speed *increases* with depth. Below the mixed layer, sea temperatures drop rapidly in an area referred to as the thermocline. In this region, temperature dominates the sound speed profile and speed decreases with depth. Finally, beneath the thermocline, the temperature becomes fairly uniform and increasing pressure causes the sound speed profile to increase with depth.

Some of the more prevalent acoustic propagation paths in the ocean include: acoustic ducting; convergence zone; bottom interaction; and shallow water propagation.

Acoustic Ducting

There are two types of acoustic ducting: surface ducts and sound channels.

Surface Ducts – The top layer of an ocean is normally well mixed and has relatively constant temperature and salinity. Because of the effect of depth (pressure), sound's speed in surface layers increases with depth. As a result, sound transmitted within this layer is refracted upward toward the surface. If sufficient energy is subsequently reflected downward from the surface, the sound can become "trapped" by a series of repeated upward refractions and downward reflections to create surface ducts or "surface channels" (see Figure 4). Surface ducts commonly form in the winter because the surface is cooled relative to deeper water; as a result, surface ducts are predictable for certain locations at specific times of the year.

Sound trapped in a surface duct can travel for relatively long distances with its maximum range of propagation dependent on the specifics of the sound speed profile, the frequency of the sound, and the reflective characteristics of the surface. As a general rule, surface duct propagation will improve as the temperature becomes more uniform and depth of the layer increases. For example, a sound's transmission is improved when windy conditions create a well-mixed surface layer or in high-latitude midwinter conditions where the mixed layer extends to several hundred feet deep.

Sound Channels - Variation of sound velocity with depth causes sound to travel in curved paths. A sound channel is a region in the water column where sound speed first decreases with depth to a minimum value, and then increases. Above the depth of minimum value, sound is refracted (bent) downward; below the depth of minimum value, sound is refracted upward. Thus, much of the sound starting in the channel is trapped, and any sound entering the channel from outside its boundaries is also trapped. This mode of propagation is called sound channel propagation. This propagation mode experiences the least transmission loss along the path, thus resulting in long-range transmission (see Figure 4).

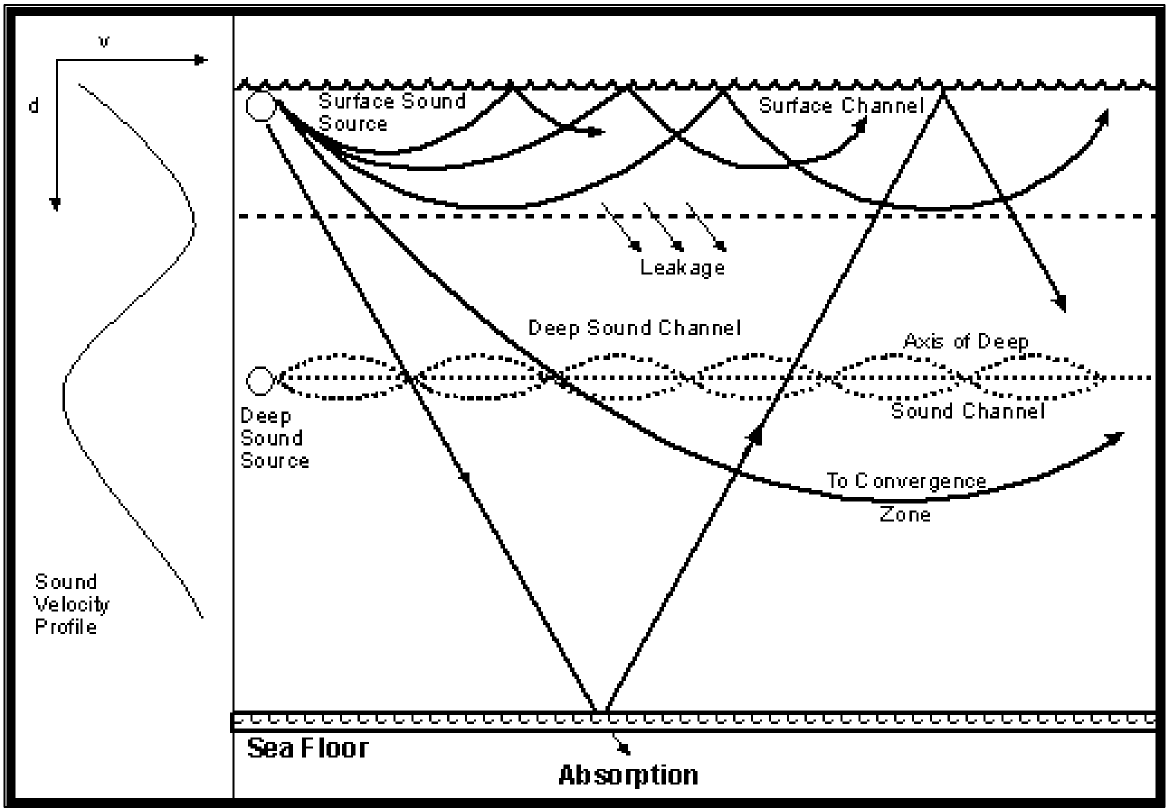

Figure 4. Typical Modes of Underwater Sound

At low and middle latitudes, the deep sound channel axis varies from 1,970 to 3,940 ft (600 to 1,200 m) below the surface. It is deepest in the subtropics and comes to the surface in the high latitudes, where sound propagates in the surface layer (see Figure 5).

Convergence Zones

The most common sound channels that LFA sonar will experience in deep water areas in summertime will produce what are known as convergence zones (see Figure 4). Convergence zones are special cases of sound-channel effects. When a surface layer is thin or when sound rays are refracted downward, regions are created at or near ocean surfaces where sound rays are focused, resulting in concentrated high sound levels. The existence of convergence zones depends on the sound speed profile and the depth of the water. Due to downward refraction at shorter ranges, sound rays leaving the near-surface region are refracted back to the surface because of the positive sound speed gradient produced by the greater pressure at deep ocean depths. These deep-refracted rays often become concentrated at or near the surface at some distance from the sound source through the combined effects of downward and upward refraction, thus causing a convergence zone. Convergence zones may exist whenever the sound speed at the ocean bottom, or at a specific depth, exceeds the sound speed at the source depth. Depth excess, also called sound speed excess, is the difference between the bottom depth and the limiting, or critical depth.

Convergence zones vary in range from approximately 18 to 36 nm (33 to 67 km), depending upon the sound speed profile. The width of the convergence zone is a result of complex set of interrelated variables; in practice, however, the width of the convergence zone is usually on the order of 5 to 10 percent of the range. For optimum tactical performance, the Navy expects to propagate SURTASS LFA signals in convergence zones in open ocean conditions.

Bottom Interaction

Reflections from the ocean bottom and refraction within the bottom can extend propagation ranges (see Figure 6). For mid- to high-level frequency sonar (greater than 1,000 Hz), only minimal energy enters into the bottom thus reflection is the predominant mechanism for energy return. However, at low frequencies, such as those used by the SURTASS LFA source, sound will penetrate the ocean floor and refraction within the seafloor, not reflection, will dominate the energy's return. Regardless of the actual transmission mode (reflection from the bottom or refraction within the bottom) this interaction is generally referred to as "bottom bounce" transmission.

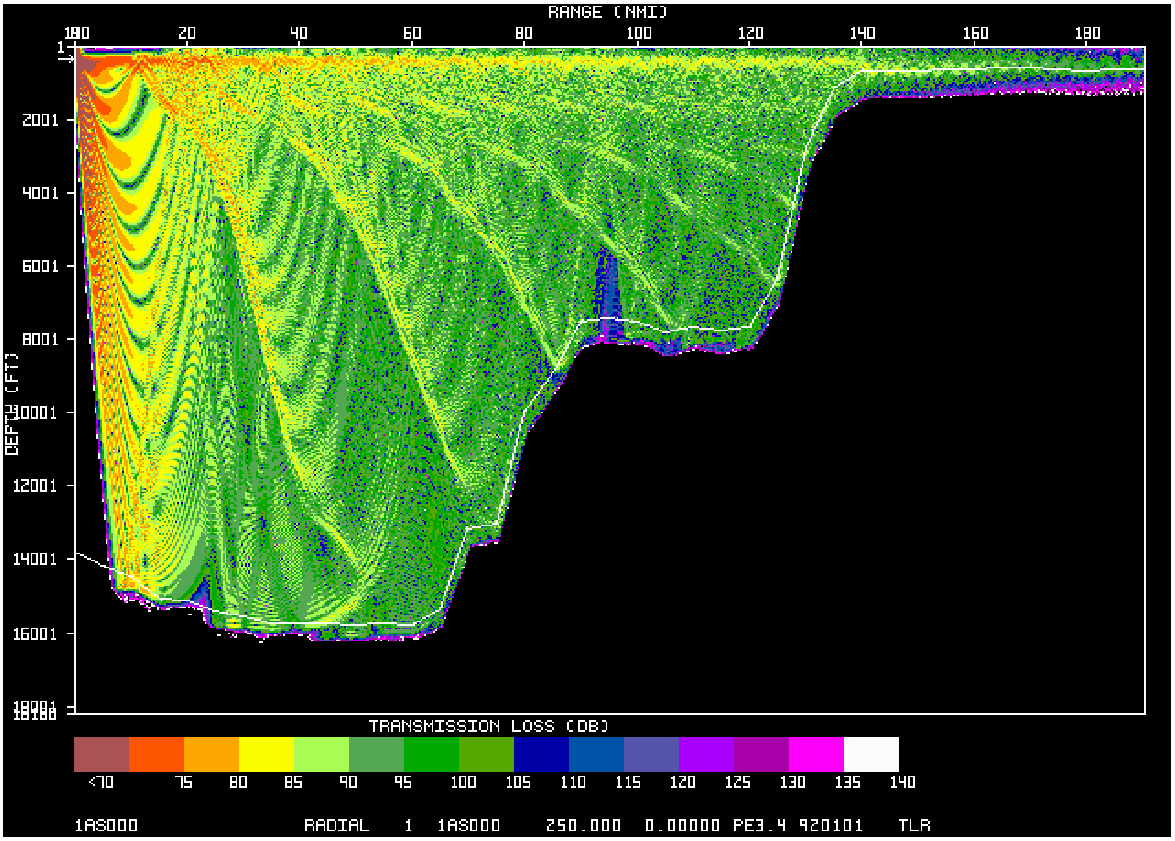

Figure 5. Plot of the sound field associated with SURTASS LFA sonar for Gulf of Alaska produced by the Navy’s parabolic equation model (which simulates transmissions losses with depth and distance). The sound source is in the upper, left corner of the plot. The colors in the plot correspond to the transmission losses presented in the color key on the bottom of the figure.

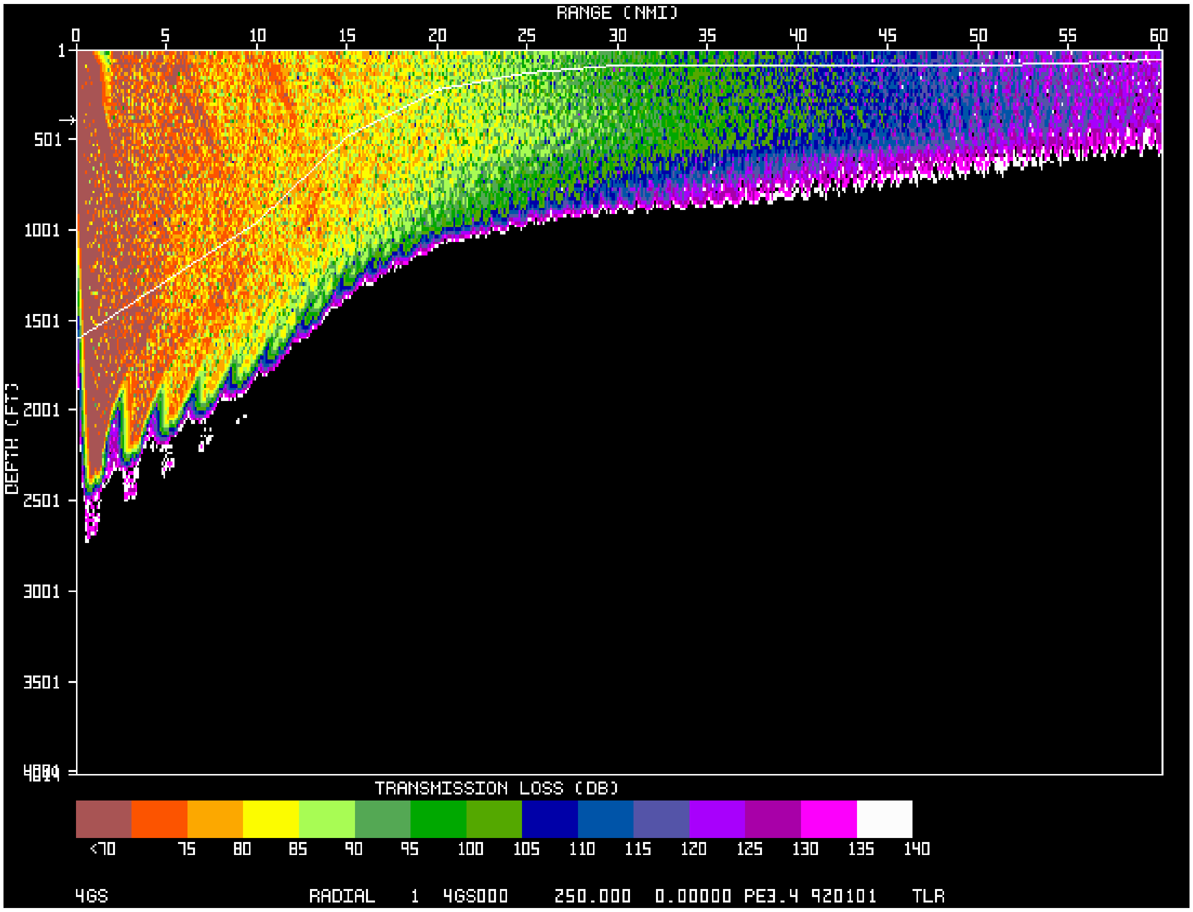

Figure 6. Plot of the sound field associated with SURTASS LFA sonar for Onslow Bay produced by the Navy's parabolic equation model (which simulates transmissions losses with depth and distance). The sound source is in the upper, left corner of the plot. The colors in the plot correspond to the transmission losses presented in the color key on the bottom of the figure. The source is in water less than 305 m (1,000 ft) deep.

Major factors affecting bottom-bounce transmission include the sound frequency, water depth, angle of incidence, bottom composition, and bottom roughness. A flat ocean bottom will produce the greatest accuracy in estimating range and bearing in the bottom-bounce mode. For LFA sonar transmissions between 100 and 330 Hz, bottom interaction would generally occur in areas of the ocean where depths are between approximately 200 m (the nominal minimum water depth for LFA sonar deployment) and 2,000 m (660 and 6,600 ft).

Shallow Water Propagation

In shallow water, propagation is usually characterized by multiple reflection paths off the sea floor and sea surface. Thus, most of the water column tends to become ensonified by these overlapping reflection paths. As LFA signals approach the shoreline, they will be affected by shoaling, experiencing high transmission losses through bottom and surface interactions. Low-frequency sound is filtered out, relative to high frequency sound, by all of this surface and bottom reflection because of its long wavelength. As a result, LFA will have little or no effect in shallow, coastal waters or the organisms that inhabit those waters.

In summary, the dominant propagation paths for SURTASS LFA signals in low and middle latitudes are convergence zone and bottom interaction (<2000 m (<6,600 ft) depth). In high latitudes, surface ducting provides the best propagation. In most open water conditions, convergence zone propagation will be most prominent. SURTASS LFA signals will interact with the ocean bottom, but those signals will not penetrate coastal waters with appreciable signal strengths because of high bottom and surface losses.

Because of spherical spreading, the 215 dB signal from a SURTASS LFA projector would be expected to attenuate by about 60 dB one kilometer from the source and by about 66 dB two kilometers from the source. In ideal oceanic conditions, a SURTASS LFA signal would lose about 120 dB to spherical spreading, so the signal would probably approach or fall below ambient levels about 960 kilometers from a SURTASS LFA source (about 600 miles).

Conditions of Exposure

This section describes the conditions under which listed species could be exposed to LFA sonar based on evaluations of the available information. This section also describes potential relationships between differing levels of exposure to LFA sonar and potential effects on listed species; therefore, they assume that the potential, biological risk of LFA sonar is a function of an animal's exposure to a sound that could adversely affect the animal's hearing, behavior, psychology, or physiology.

Richardson *et al*. (1995) argued that it was important to estimate the radius within which acoustic effects would be expected to assess the potential effects of man-made noise on marine mammals. They proposed four zones of potential influence: (a) the most extensive of these zones is the zone of audibility, within which an animal might hear a sound; (b) the zone of responsiveness is the region within which an animal might react behaviorally or physiologically to a sound. This zone can be smaller than the zone of audibility, since marine mammals often do not react overtly to noises that are faint but presumably audible; (c) the zone of masking is the region within which noise is strong enough to interfere with an animal's ability to detect other sounds, like communication or echolocation calls, prey sounds, or other natural sounds that provide information about the animal's environment. The zone of masking is highly variable in size; and (d) the zone of hearing loss, discomfort, or injury is the area near a noise source where, for explosions and other strong sources, received sound levels are high enough to cause discomfort or tissue damage to auditory or other systems.

Richardson *et al*. (1995) argued that man-made sounds with received levels slightly less than ambient (in the corresponding band or a signal to noise ratio of <0 dB) are not likely to disturb an animal even though the sounds were audible, although there is little information to validate this point. For example, gray whales that were swimming toward a source of killer whale sounds first reacted to the sound when the received level was similar to ambient; that is, when the signal to noise ratio was ≥0 dB (Malme *et al*. 1983). If gray whales do not appear to react to sounds of potential predators unless signal to noise ratios are ≥0 dB, they are not likely to react to man-made sounds with signal to noise ratios <0 dB. Therefore, animals that are exposed to received levels that are slightly less than ambient are not likely to elicit responses in marine mammals.

Intensity

As discussed previously in *Propagation*, sound transmissions are usually measured in terms of sound pressure levels, which are denoted as decibels and which have a reference pressure value of 1 microPascal (μPa). The logarithmic nature of the decibel (dB) 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). Humans perceive a 10 dB increase in noise as a doubling of sound level, or a 10 dB decrease in noise as a halving of sound level.

Inside the LFA mitigation zone during a ping, a marine mammal could be exposed to sound levels at or above 180 dB and could experience permanent threshold shifts or other injury. However, the LFA mitigation zone was established and designed to prevent marine mammal or sea turtles from being exposed to these energy levels. Given the size of the LFA mitigation zone (extending to approximately 0.75 to 1.00 km [0.40 to 0.56 nm] from the transmitter) and the additional 1-km buffer zone, the detection probabilities associated with the High Frequency Marine Mammal Monitoring sonar (above 95 percent probability of detecting small dolphins at about 750 m [0.4 nm], whale calves at 1,000 m [0.56 nm] and large whales at more than 1,500 m [0.81 nm]), and the depth of the transmitters, a marine mammal would have a high probability of being detected within the LFA mitigation zone and, as a result, a low probability of being exposed to sound levels greater than 180 dB.

For this exposure to occur, the cetacean would have to occur in the same approximately 4-kilometer wide water column as the transmitter, would have to enter the LFA mitigation zone without being detected, and would have to remain in the LFA mitigation zone when the LFA transmitter was operating. Based on the available information, we believe the probability of all of these events occurring, although possible, is extremely improbable.

Further, SURTASS LFA will be operated to ensure that sonar sound fields do not exceed 180 dB (re 1 μPa_{rms}) within 12 nautical miles (22 kilometers) of any coastline, including offshore islands, or designated offshore areas that are biologically important for marine mammals outside the 12 nautical mile (22 kilometer) zone during seasons specified for a particular area (see Table

1). When in the vicinity of known recreational and commercial dive sites, SURTASS LFA sonar would be operated to ensure that the sound field at these sites would not exceed 145 dB, adding an additional level of protection for marine mammals located in dive sites.

As mentioned in the Description of the Proposed Action section of this opinion, NOAA's Office of National Marine Sanctuaries has recommended the following protective measures to the Navy for SURTASS LFA sonar operations: (1) in the Monterey National Marine Sanctuary, received levels should not exceed 180 dB throughout the sanctuary; (2) in the Gulf of the Farallones and Cordell Bank National Marine Sanctuaries, received levels should not exceed 180 dB for areas that extend beyond 12 nm (22 km); (3) in the Olympic Coast National Marine Sanctuary, received levels in the National Marine Sanctuary should not exceed 180 dB from the area from the shore to 23 nm (37.4 km) in the months of December, January, March and May; and (4) in the Hawaiian Islands Humpback Whale National Marine Sanctuary, received levels should not exceed 180 dB from November through May 1.

These constraints prevent most threatened and endangered species of whales from being exposed to SURTASS LFA sonar at sound pressure levels exceeding 180 dB in areas that are critical to their ecology, critical to large portions of their populations, or both. For example, these constraints prevent right whales in the western Atlantic Ocean and gray whales in the eastern Pacific Ocean from being exposed to SURTASS LFA sonar signals exceeding 180 dB in the critical feeding and rearing areas and migratory pathways that have been identified for them. These constraints also prevent humpback whales from being exposed to SURTASS LFA signals exceeding 180 dB in their wintering grounds in Hawaii, summer feeding grounds in the Gulf of Alaska and Glacier Bay; and blue, fin, humpback, bowhead, and sperm whales from being exposed to SURTASS LFA signals at those sound pressure levels in critical feeding grounds in the Arctic and Antarctic areas.

The Navy's acoustic modeling also provides insights into potential exposure of marine mammals to SURTASS LFA sonar (see Final EIS, Subchapter 4.2.1) to exposure levels below 180 dB, which may result in significant changes in biologically important behavior. Specifically, the Navy developed the Acoustic Integration Model (AIM) to simulate acoustic exposures during a hypothetical SURTASS LFA sonar operation by simulating different "acoustic modeling sites." The result of these simulations projected the percentage of marine mammal populations that could be affected by SURTASS LFA sonar under different scenarios. These sites were chosen to represent conditions that would allow the Navy to model the highest potential for effects from the use of SURTASS LFA sonar and, as a result, probably represent worst case scenarios. These scenarios included areas close to land (where biological densities are higher), best sound propagation conditions for the area (which would not always occur), and season of highest marine mammal density (areas the Navy would routinely avoid because of the potential for excessive shutdowns).

Because the Navy does not propose to employ more than two SURTASS LFA sonar vessels during the next five-year period, the percentages of marine mammal stocks depicted as examples in Table 4 (see also Table 4.2-11 and 4.2-12 of the Final EIS and section 4.2 of the EIS for a more extensive explanation of the computations) overestimate the risk of exposure because they are based on 24 missions, not the 12 missions (6 per vessel) that are expected to occur with two vessels. Considering that SURTASS LFA sonar missions are more likely to occur in the open ocean rather than in coastal waters, and that the LOA process will identify areas based on biologically-sensitive seasons, NMFS believes the actual percentages of taking by harassment incidental to SURTASS LFA sonar are significantly lower than the results shown in Table 4.

Table 4. Examples of annual estimates of the percentage of different marine mammal populations that would be potentially exposed to LFA sonar in different regions

Percentage of the population that could be exposed to sound levels <180 dB

Region	Eastern North Pacific	Western North Pacific	South Pacific	Indian Ocean
Blue Whale	8.36	6.27	0.32	Not modeled
Fin Whale	1.03	1.07 (0.03)	0.29	Not modeled
Sei Whale	Not modeled	Not modeled	0.16	Not modeled
Humpback Whale	2.58	3.29 (0.21)	4.44	0.20
Gray Whale	3.43	5.30	Not modeled	Not applicable
Right Whale (n)	4.13	Not modeled	Not modeled	Not modeled
Right Whale (s)	Not modeled	Not modeled	1.38	Not modeled
Sperm Whale	0.16	Not modeled	0.32	0.03
Steller sea lion	4.53	0.19	Not applicable	Not applicable
Hawaiian monk seal	2.39	Not applicable	Not applicable	Not applicable
Region	**Eastern North Atlantic**	**Western North Atlantic**	**South Atlantic**	**Mediterranean Sea**
Blue Whale	16.39	16.06	0.85	Not modeled
Fin Whale	0.64	1.77	0.41	7.69
Sei Whale	3.92	5.54	Not modeled	Not modeled
Humpback Whale	3.12	7.12	1.80	Not modeled
Right Whale (n)	Not modeled	2.52	Not modeled	Not modeled
Sperm Whale	0.41	Not modeled	Not modeled	13.40

Duration

As discussed previously, at-sea missions of the SURTASS LFA sonar system would nominally consist of two 9-day exercise segments over a 30-day period. Active sonar operations could be conducted up to 20 hrs during an exercise day, although the system would actually transmit for a maximum of about 4 hours per day. The duration of a typical SURTASS LFA ping would range from 6 to 100 seconds, with no more than 10 seconds at a single frequency; intervals between pings would range from 6 to 15 minutes. Pings would consist of various signal types that vary in frequency (between 100 and 330 Hz) and duration (including continuous wave and frequency-modulated signals). When the system is turned off, no additional energy would enter the ocean's environment. Each SURTASS LFA sonar system, with a maximum duty cycle of 20 percent, could actively transmit for 432 hours per year.

The duration of an animal's exposure to SURTASS LFA signals will depend on their proximity to the transmitter and their location in the water column. Nevertheless, because of the length of individual pings, individual animals would be exposed to SURTASS LFA transmissions for periods ranging from 6 to 100 seconds if the animals were swimming in a convergence zone.

Frequency

Individual animals or groups of animals have a low statistical probability of being exposed to SURTASS LFA sonar signals on several, separate occasions. However, the number of times an animal would be exposed to sound pressure levels associated with SURTASS LFA transmissions will depend on the deployment schedule of the two vessels.

There are no published data on marine mammals regarding responses to repeated exposure to low frequency sound. Evidence from studies of repeated exposure to other impulsive sounds suggest that the risk threshold is lowered by 5 dB per ten-fold increase in the number of pulses per exposure if the number of pulses per exposure is less than 100 (Richardson *et al*. 1995, citing Kryter 1985). These findings are consistent with qualitative statements by Crocker (1997).

The Navy represented the probability of risk, using stressor-response functions generated by mathematical simulation. These functions, which are represented as cumulative probability distributions or cumulative distribution functions, have values near zero at very low exposures, and values near one for very high exposures. From this distribution function, received levels of 150 dB have a 2.5 percent likelihood of significantly changing behavior that is biologically important to marine mammals. Received levels corresponding to a 50 percent risk on this curve is 165 dB. However, at 180 dB, the risk of significant change in a biologically important behavior is 95 percent.

One of the Navy's simulations assumed that a marine mammal was exposed to a total of ten SURTASS LFA sonar transmissions, or pings, at received levels between 150-159 dB. The simulation showed that the animal was exposed to two pings at 150 dB received level, none at 151 dB received level, three pings at 152 dB received level, etc. To arrive at a total single-ping equivalent for the entire exposure, the Navy first calculated the intensity level for each ping (i.e., 1×10^{15} μPa for each of the two 150 dB received level exposures, 1.58×10^{15} μPa for each of three 152 dB received level exposures, etc.). These intensity values were then squared and added together; the estimates concluded that these different pings resulted in a total single ping equivalent of 160.47 dB (see Navy 2001, 4.2.3.1. "Effects of Repeated Exposure"). In this example, the risk function would predict a 24.48 percent probability of significant change in behavior that is biologically important to an animal.

Mitigation Measures to Minimize the Likelihood of Exposure

The Navy proposes to use a monitoring program to avoid potentially exposing marine mammals to LFA transmissions at high decibel levels. As discussed in the Description of the Proposed Action, this monitoring program includes visual, passive acoustic, and active acoustic monitoring of a 180 dB mitigation zone and an additional 1 km buffer zone.

The effectiveness of visual monitoring is limited to daylight hours, and its effectiveness declines during high sea states. In line transect surveys, the range of visual sighting effectiveness (distance from the ship's track, called effective strip width) varies with an animal's size, group size, reliability of conspicuous behaviors (blows), pattern of surfacing behavior, and positions of the observers (which includes the observer's height above the water surface). For most large baleen whales, effective strip width can be about 3 km (1.6 nm) up through Beaufort 6 (Buckland *et al*. 1993). For harbor porpoises the effective strip width is about 250 m (273 yd), because they are much smaller and less demonstrative on the surface than the baleen whales (Palka 1996). The percentage of animals that will pass unseen is difficult to determine, but for minke whales, Schweder *et al*. (1992) estimated that visual survey crews did not detect about half of the animals in a strip width. Palka (1996) and Barlow (1988) estimated that visual survey teams did not detect about 25 percent of the harbor porpoises in a strip width.

The effectiveness of passive acoustic detection is considered to be higher than visual monitoring. Thomas *et al*. (1986) and Clark and Fristrup (1997) concluded that the effective strip width and detection rates for passive acoustic monitoring is greater than that for visual, but the percentage of animals that will be undetected by the methods is unknown. Frequency coverage for this mitigation method using the SURTASS passive array is between 0 and 500 Hz, so vocalizing animals are more likely to be detected than animals that do not vocalize. This would increase the detection rate of gray, humpback, fin, blue, and minke whales, and some of the beaked whale and dolphin species.

The HF/M3 sonar is the final measure the Navy proposes to use to detect animals within 1 to 2 kilometers of the projectors. Recent testing of the HF/M3 sonar demonstrated a probability of single-ping detection above 95 percent within the LFA mitigation zone for most marine mammals (see the Navy's Final EIS, Navy 2001, section 2.3.2.2). If any of these monitoring methods detects animals within this zone, the projectors would be shut down until the animal(s) move out of the mitigation zone. Combined with the visual monitoring and passive acoustic monitoring protocols, this should minimize the risk of marine mammals being exposed to sound pressure levels in excess of 180 dB.

Exposure profiles

Exposure profiles summarize how exposure occurs; what species would be exposed[5]; how much exposure would occur; when and where exposure would occur; if exposure would be expected to vary with the biology and ecology of the species or characteristics of the environment; the likelihood of exposure occurring; and uncertainty associated with these summaries (U.S. EPA 1998).

As discussed previously, an animal would be exposed to a SURTASS LFA sonar signal during a ping, which would last between 6 and 100 seconds (nominally 60 seconds). The potential effects

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[5] EPA's risk assessment guidelines use the term "ecological entity" in its definition of "exposure profile." Ecological entity is a general term that refer to a species, a group of species, an ecosystem function or characteristic, or a specific habitat that are components of an assessment endpoint. Ecological entities in biological opinions are always threatened or endangered species or critical habitat that has been designated for them.

of these pings decrease with distance from the projectors; animals that occur within a column of water within 1,000 meters of these projectors during one of these pings would have the greatest risk of injury, although the probability of such exposure seems small given the short duration of a ping and the Navy's monitoring protocols. However, the Navy's ability to detect animals using these monitoring protocols may vary with the size of the animal: endangered whales, leatherback turtles, and schools of salmon have the greatest probability of being detected by the monitoring systems associated with the SURTASS LFA sonar systems, which gives them the lowest probability of being exposed to signals at received levels approximating 180 dB. The Navy's ability to detect pinnipeds, smaller turtles, and small groups of salmon remains uncertain.

Because of the operating constraints that have been placed on employment of SURTASS LFA, animals with offshore distributions have a higher risk of being exposed to SURTASS LFA sonar signals or of being exposed to higher received levels than animals with coastal distributions. Within the ocean's surface layer (uppermost 150 feet) the sound field would be dominated by sound generated by wave action, rain, and other surface activity; which would mask sounds generated by SURTASS LFA sonar. Animals with an offshore distribution that occur in this surface layer have a lower risk of exposure to coherent LFA sonar signals, although they may be exposed to incoherent energy associated with this signals.

Below this surface layer and outside of the SURTASS LFA mitigation zone, sound pressure levels from SURTASS LFA transmissions would depend largely on physical, oceanic conditions, and the Navy's use of surface ducts, convergence zones, and sound channels, which would determine whether the transmission losses were dominated by spherical or cylindrical spreading models. In convergence zones, SURTASS LFA signals would resurface from bottom layers at 33 to 67 km intervals (18 to 36 nm), depending upon the signal's sound speed profile. Between convergence zones, animals would not be exposed to coherent SURTASS LFA signals, but could be exposed to incoherent energy associated with those signals. Therefore, animals with offshore distributions have a higher risk of exposure to SURTASS LFA sonar transmissions if they occur in the LFA mitigation zone, if they dove to depths that might expose them to a surface duct or sound channel, or if they occurred in a convergence zone. In any of these scenarios, an animal would be exposed to SURTASS LFA signals for the 6 to 100 second duration of a single ping.

Steller sea lions. About 10 percent of the Steller sea lion population in the eastern Pacific Ocean have a chance of being exposed to SURTASS LFA transmissions. Because of the restrictions that prevent the SURTASS system from being operating within the 200-meter isobath of the United States, sea lions in the Gulf of Alaska, eastern Alaskan peninsula, Kodiak Island area, and the eastern population are less likely to be exposed to these transmissions. Animals occupying rookeries and haulouts in the Bering Sea and along the northern coast of the Alaskan peninsula, which includes some of the largest rookeries in the western population, are not likely to be exposed. Sea lions on rookeries and haulouts in the Aleutian Island chain, which currently support the smallest and most vulnerable proportion of the western population, have the highest risk of exposure to SURTASS LFA transmissions. Based on the Navy's models, these animals would receive levels ranging from slightly less than 160 dB to less than 120 dB with a unimodal distribution centered midway between 150 and 145 dB tapering quickly to 130 dB.

Guadalupe fur seals. These fur seals could be exposed to SURTASS LFA transmissions throughout their distribution from Guadalupe Island north to the Channel Islands. Based on the results of the Navy's models, about 0.45 percent of these fur seals have a chance of being exposed to SURTASS LFA transmissions at received levels less than 180 dB (these data are from simulations for Site 8, which represents the waters surrounding San Nicholas Island, California). Of the animals that are exposed to these transmissions, received levels would range between 120 dB (which is generally considered ambient in marine ecosystems) and 160 dB.

Hawaiian monk seals. Although Hawaiian monk seals generally reside in coastal waters near haul-out areas, they forage in deep water and dive to at least 490 m (1,608 ft; Reeves *et al.* 1992), which could expose them to low frequency sounds from SURTASS LFA. The Navy's simulation models suggest that Hawaiian monk seals would be exposed to SURTASS LFA transmissions northwest of the Island of Kauai, where 2.39 percent of the population might be exposed to SURTASS LFA transmissions (these data are from simulations for Sites 6, 7, and 12, which represent the waters surrounding the Islands of Kauai and Oahu in Hawaii). Nevertheless, monk seals exposed to these transmissions, received levels would range between 130 dB and 160 dB.

Blue whales. Based on the results of the Navy's models, blue whales had the second highest probability of being exposed to SURTASS LFA sonar (Table 4): about 16 percent of the blue whale population in the North Atlantic Ocean and 6 to 8 percent in the North Pacific Ocean could be exposed to SURTASS LFA sonar depending upon how SURTASS LFA vessels were deployed. The highest proportions of blue whales would be exposed off northern Europe, particularly in the Denmark Strait (Site 17) and off Bergen, Norway (Site 18), which were responsible for 13.77 and 9.83 percent of the exposure, respectively (Note: this analysis was based on the four vessels that were projected to be deployed instead of the 2 vessels that will actually be deployed). The lowest proportions of the exposure were off the coasts of South America. Blue whales in the Southern Ocean (the Antarctic Convergence Zone, which is an offshore biologically important area) are not likely to be exposed to LFA signals at any detectable level unless these animals migrate out of the Southern Ocean.

The intensities of exposure would vary with geographic location. In the Denmark Straits and Bergen, Norway regions, blue whales would be exposed to received levels ranging from about 120 dB to about 170 dB, with the highest proportion exposed to received levels between 150 and 160 dB. In the north Gulf of Alaska, for example, blue whales could be exposed to received levels ranging from about 130 dB to 165 dB, with a unimodal distribution centered between 145 and 150 dB.

Bowhead whales. Because of the operational restrictions associated with SURTASS LFA, the only bowhead whales that are at risk of being exposed to SURTASS LFA transmissions are the Sea of Okhotsk and Spitsbergen populations. Although the Navy did not model potential exposure of bowhead whales, we used the exposure risk of the western Pacific gray whales, about 5 percent, as an index of the proportion of the Sea of Okhostk bowhead whale population that could be exposed to SURTASS LFA transmissions. Similarly, we have no specific information on received levels for bowhead whales, but received levels for gray whales ranged from slightly less than 150 dB to less than 120 dB with a unimodal distribution peaking at slightly less than 145 dB, quickly tapering to 130 dB.

Gray whales. About 5 percent of the western Pacific gray whale population has a chance of being exposed to SURTASS LFA transmissions depending upon how the vessels are deployed. Based on the results of the Navy's models, the western Pacific population of gray whales had the highest probability of being exposed to SURTASS LFA sonar south of Japan. The coastal migratory habit of gray whales would protect them from being exposed to sound pressure levels greater than 180 dB. In the north Pacific Ocean, received levels for gray whales (model runs for the eastern Pacific population of gray whales, which are not listed) ranged from slightly less than 150 dB to less than 120 dB with a unimodal distribution peaking at slightly less than 145 dB, quickly tapering to 130 dB.

Fin whales. Based on the results of the Navy's models, fin whales had the highest risk of being exposed to SURTASS LFA sonar in the Straits of Sicily, in the Mediterranean Sea, where about 8 percent of the fin whale population could be exposed to SURTASS LFA transmissions (Table 4). Elsewhere in the Atlantic and Pacific Oceans, about 1 percent of the fin whales had a risk of being exposed to SURTASS LFA transmissions, although the percentage was considerably less than 1.0 at most sites. In the Atlantic Ocean, received levels for fin whales ranged from slightly less than 170 dB to less than 120 dB with a bimodal distribution centered between 135 and 140 dB, quickly tapering to 170 dB.

Humpback whales. Based on the results of the Navy's models, humpback whales had the highest risk of being exposed to SURTASS LFA sonar in the Gulf of Alaska, where about 12 percent of the population could be exposed to SURTASS LFA transmissions. Humpback whales faced their second highest risk of exposure to those transmissions in Denmark Strait region, where 5.33 percent of the population could be exposed to SURTASS LFA transmissions. Elsewhere in the Pacific and eastern North Atlantic Ocean, about 3 to 4 percent of the humpback whales had a chance of being exposed to SURTASS LFA transmissions. In the north Pacific Ocean, received levels for humpback whales ranged from slightly less than 160 dB to less than 120 dB with a unimodal distribution centered between 145 and 150 dB, quickly tapering to 130 dB.

Right whales. About 4 percent of the right whales in the eastern north Pacific and about 3 percent of the right whales in the western north Atlantic Ocean have a chance of being exposed to SURTASS LFA transmissions (again depending on how the vessels are deployed). Because of the coastal habit of right whales, the restrictions that limit SURTASS LFA transmissions to less than 180 dB within the 200-meter isobath, and transmission losses, few of the right whales along the Atlantic coast are likely to be exposed to received levels above ambient. received levels ranged from about 170 dB to less than 120 dB with the distribution peaking between 125 and 130 dB and tapering off at a regular rate to 170 dB.

Although the specific distribution of Pacific right whales is largely unknown, these animals are believed to have the same affiliation with coastal areas as right whales in the Atlantic; however, Pacific right whale on foraging grounds off Bristol Bay in the eastern Bering Sea are not likely to be exposed to SURTASS LFA transmissions. Less than 2 percent of the right whales in the southern Pacific had a chance of being exposed to these transmissions. Their received levels ranged from about 148 dB to less than 120 dB with a unimodal distribution centered at slightly less than 145 dB (which quickly tapered off to 130 dB).

Sei whales. Based on the results of the Navy's models, sei whales had the highest risk of being exposed to SURTASS LFA sonar off Sable Island Banks (east of Nova Scotia), where about 3.25 percent of the population could be exposed to SURTASS LFA transmissions. Humpback whales had their second highest risk of exposure to those transmissions in Denmark Strait region and Northeast Norwegian Basin, where 2.1 and 2.5 percent of the population, respectively, could be exposed to SURTASS LFA transmissions. Elsewhere in the Atlantic Ocean, about 4 to 5 percent, and less than 1 percent of the sei whales in the Pacific Ocean had a chance of being exposed to SURTASS LFA transmissions. Received levels for sei whales in the western north Atlantic ranged from slightly more than 170 dB to less than 120 dB with a distribution peaking between 125 and 140 dB.

Sperm whales. Based on the results of the Navy's models, sperm whales had the highest risk of being exposed to SURTASS LFA sonar in the Mediterranean Sea, particularly the Straits of Sicily and Leventine Sea, where about 5 and 3 percent of the population, respectively, could be exposed to SURTASS LFA transmissions. Elsewhere in the Pacific and eastern North Atlantic Ocean, less than 1.0 percent of the sperm whales had a chance of being exposed to SURTASS LFA transmissions. Received levels for sperm whales ranged from slightly more than 160 dB to about 120 dB in the both regions, although a larger percentage of sperm whales in the Straits of Sicily would be exposed to higher received levels.

Sea turtles. The Navy did not simulate potential exposure of sea turtles to SURTASS LFA transmissions. Because of their ecology, only the juvenile and adult stages of sea turtles could be potentially exposed to SURTASS LFA transmissions. We assume that the monitoring protocols associated with SURTASS LFA would be more effective with larger sea turtles, like adult leatherback and loggerhead turtles, than with species like olive ridley, Kemp's ridley, smaller leatherback, hawksbill, green, and some loggerhead turtles; the monitoring protocols may not detect some individual members of these species at all, which would increase their risk of exposure to sound pressure levels associated with SURTASS LFA within the mitigation zone (that is, ≥ 180 dB) if they encountered SURTASS LFA vessels during a ping.

Although the probability of an interaction between SURTASS LFA sonar and individuals of any of these species is statistically small (the Navy's analyses concluded that the possible number of times a leatherback sea turtle could be in the vicinity of a SURTASS LFA sonar vessel would be less than three out of 18,000 animals per year per vessel; with the monitoring protocols, the Navy concluded that this number would approach zero), this probability assumes that the sea turtles and SURTASS LFA vessels would be randomly distributed in the ocean (in such large space, two randomly-distributed objects would have a low probability of co-occurring). In practice, both sea turtles and SURTASS LFA vessels would not be randomly distributed in the ocean, which would increase a turtle's likelihood of being exposed to LFA transmissions.

Salmon. The Navy did not simulate potential exposure of Atlantic or Pacific salmon to SURTASS LFA transmissions. Because of their ecology, only the marine life stages of these salmon could be potentially exposed to SURTASS LFA transmissions depending on deployments. In the Atlantic Ocean, listed salmon have a risk of being exposed to deployments in the Gulf of Maine, in marine waters off Newfoundland, and in the southern portion of the Labrador Sea (because of operational restrictions, they would not be exposed throughout the entire Labrador Sea). In the Pacific Ocean listed salmon have a risk being exposed to deployments north of about 40º North latitude. Although salmon generally occur near the surface (within 8 to 10 meters of the surface), sockeye salmon have been caught at depths up to 61 meters while chinook salmon have been caught at depths up to 110 meters. Since the HF/M3 sonar is patterned after technology commercial and sports-fishing industries use to locate fish, we will assume that the sonar would locate schooling species like Pacific salmon, which would minimize their likelihood of being exposed to sound pressure levels in excess of 180 dB.

Response Analysis

For species that are likely to be exposed to SURTASS LFA sonar transmission, this response analysis examines the relationship between received levels and ecological effects (generally called “stressor-response analyses”); the plausibility of adverse effects occurring as a result of exposure; and linkages between measurable ecological effects and vital rates or biologically-important behavior in populations of listed species, which are the endpoints for this assessment. In this analysis we consider potential injury, resonance effects, stranding, masking, and biologically-important behavioral responses measures of the effects of exposure to SURTASS LFA sonar. Specifically, we evaluate the available evidence to determine if SURTASS LFA sonar can be expected to (1) physically injure marine species in a way that would have acute or chronic effects or (2) elicit behavioral responses that would have longer-term, chronic effects on the viability of populations of a species (as discussed previously, we initially screened for indirect effects resulting from the disruption of food chains or interactions between species, but our searches produced no evidence of these phenomena). Although this section of our assessment included concerns for effects on individual animals, our assessment focused on the probable effects of SURTASS LFA sonar on populations and, through populations, listed species.

To establish relationships between stressor levels and their effects on threatened and endangered species, our assessment relied on the Navy’s acoustic models and risk continuum analyses (see U.S. Navy 1999b, 2001a, 2001b), which estimate the risk of injury and changes to biologically important behavior that might be caused by SURTASS LFA sonar and the Navy’s analyses correlating the risk of biologically significant behavior to received levels (single ping equivalents in decibels) using probability distribution functions. The results of the Navy’s analyses are presented as continuous functions that are analogous to the stressor-response curves, low received levels would not be expected to elicit a negative response in a species; at the other end of these curves, high received levels would be expected to elicit much more serious, negative responses.

To assess the plausibility of adverse effects resulting from exposure to SURTASS LFA sonar, we evaluated various lines of evidence from published and unpublished sources on the effects of SURTASS LFA sonar on threatened and endangered species, other studies of low frequency sound, and reports on the effects of other sonars. We evaluate the same lines of evidence to establish linkages between the potential effects of SURTASS LFA sonar on biologically important behavior of listed species exposed to the sonar.

Given that all of the threatened and endangered species of whales, sea lions, sea turtles, and Pacific salmon have some risk of being exposed to LFA sonar signals at some sound pressure levels, the susceptibility of these species is relevant to this assessment of effects. As discussed in the environmental impact statement on the SURTASS LFA sonar (Navy 2001a), the Navy's Scientific Research Program conducted field studies of the types of responses of whales to SURTASS LFA sonar signals and how those responses scaled relative to received level and environmental setting. Before the research program, marine mammal scientists expected obvious responses from whales at exposure levels greater than 140 dB and statistically significant responses at levels around 120 dB, based on responses detected in previous research to continuous industrial sounds (Malme *et al*. 1983, 1984; Richardson *et al*. 1995b). The results of the Scientific Research Program did not support these expectations since no significant behavioral responses were noted for received levels up to 155 dB.

The LFS Scientific Research Program was designed to ensure that the operation of SURTASS LFA sonar did not expose any marine mammal to received levels exceeding 160 dB (Navy 1999b). The investigators associated with this research program explicitly focused on situations that promoted high received levels (maximum 160 dB), but were seldom able to achieve received levels in the high region of this exposure range due to the natural movements of the whales and maneuvering constraints of the low frequency source vessel. However, the research program produced new information about responses to the SURTASS LFA sonar sounds at received levels from 120 to 155 dB.

During the first phase of the research program, the source ship operated routinely with the full LFA array (18 source projectors) at source levels similar to those that would be used in normal Navy operations. The ship also approached whales while operating two of the projectors at full power levels. Over the 19-day period, there were no immediately obvious responses from either blue or fin whales as noted during observations made from any of the research vessels during playback of LFA sounds.

An earlier study examined responses of gray whales migrating along the California coast to various sound sources located in their migration corridor (Malme *et al*. 1983, 1984). Gray whales 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. This study was replicated in Phase II of the LFS Scientific Research Program using SURTASS LFA sonar stimuli. However, the Phase II research demonstrated that it may be invalid to apply the inshore (2 km from shore) response model (when 50 percent of the whales avoided SURTASS LFA sonar stimuli at received levels of 141 ±3 dB) to sources that are offshore (4 km from shore) of migrating whales, and that whales did not avoid offshore sources at received levels of 140 dB. This implies that the inshore avoidance model _ in which 50 percent of the whales avoid exposure to levels of 141 ±3 dB _ may not be valid for whales in proximity to an offshore source (Buck and Tyack 2000). It should also be noted that this research was conducted between 2 and 4 km from shore, but SURTASS LFA sonar will not operate within 12 nm of any coastline.

The third phase of the research program examined potential effects of SURTASS LFA sonar transmissions on singing humpback whales. These whales showed some apparent avoidance responses and cessation of song during specific LFA sound transmissions at received levels ranging from 120 to 150 dB. However, an equal number of singing whales exposed to the same levels showed no cessation of song during the same LFA sound transmissions. Of the whales that did stop singing, there was little response to subsequent LFA sound transmissions; most joined with other whales or resumed singing within less then an hour of the possible response. Those that did not stop singing, sang longer songs during the period of LFA transmissions, and returned to baseline after transmissions stopped (Miller *et al*., 2000). Further analysis is required to establish how often male humpbacks stop singing in the absence of the SURTASS LFA sonar transmissions, and to evaluate the significance of the song cessation observed during playbacks.

These experiments, which exposed baleen whales to received levels ranging from 120 to about 155 dB, detected only minor, short-term behavioral responses. Short-term behavioral responses do not necessarily constitute significant changes in biologically important behaviors. The fact that none of the Scientific Research Program observations revealed a significant change in a biologically important behavior helped determine an upper bound for risk. The Scientific Research Program results cannot, however, be used to prove that there is zero risk at these levels. Accordingly, the risk continuum approach assumes that risk is small, but not zero, at the received levels achieved during the Scientific Research Program. The risk continuum modeled a smooth increase in risk that culminates in a 95 percent level of risk of significant change in a biologically important behavior at 180 dB.

The risk curves produced from data from the Navy's research program and the modeling analyses suggests the probability of significant changes in biologically important behavioral between marine mammal responses and SURTASS LFA sonar at various received levels. However, reports of the potential responses of marine mammals to SURTASS LFA sonar would imply different ranges of responses to different received levels. SURTASS LFA sonar has been reported as eliciting behavioral responses that have no obvious relationships to the survival, reproductive success, or longevity of individual members of numerous marine mammal species that include the threatened or endangered species being considered in this opinion.

Specific responses of whales to underwater sound transmissions have been the subject of three separate studies (Bowles *et al*. 1994, Croll *et al*. 2001, Miller *et al*. 2000, U.S. Navy 2000b). These studies identified increases in the length of whale songs (Miller *et al*. 2000), and changes in respiration and reorientation rates (Bowles *et al*. 1994). Croll *et al*. (2001) did not report responses that were statistically or biologically significant.

Potential Injury

As discussed previously, at-sea missions of the SURTASS LFA sonar system would normally consist of two 9-day exercise segments over a 30-day period. Active sonar operations could be conducted up to 20 hrs during an exercise day, although the system would actually transmit for a maximum of about 4 hours per day. A typical transmission or "ping" would consist of various signal types that vary in frequency (between 100 and 330 Hz) and duration (including continuous wave and frequency-modulated signals). Pings would range in duration from between 6 and 100 seconds, with no more than 10 seconds at a single frequency; the interval between pings would range from 6 to 15 minutes.

The center of LFA arrays would operate at a depths of 400 feet (122 m), which would generally place the array and the main, sound transmission beam produced by the arrays below the mixed-layer depth of an ocean. The transmitted signal would move in an extended wave through the ocean, initially moving away from the surface, then rising at some distance from the source, before sinking again (in some instances affected by ducting, sound channels, convergence zones, and bottom interactions). At the source, an LFA signal is approximately 215 dB; this signal would attenuate with time and distance from the source. Received levels from the array attenuate to 180 dB nominally from 750 to 1,000 m (2,460 to 3,281 ft).

SURTASS LFA sonar has the greatest *potential* for injuring threatened or endangered species if an animal is close the one or more of the projectors during a ping. If that occurred, the animal could be exposed to sound pressure levels between 215 and 180 dB, which *could* injure listed species by causing permanent threshold shifts (PTS). To minimize the likelihood of this occurring, the Navy proposes to use HF/M3 sonar, visual monitoring, and passive acoustic monitoring, to detect animals within 1 to 2 kilometers of the projectors; if any of these monitoring methods detects animals within this zone, the projectors would be shut down until the animal(s) moved out of the zone. Recent testing of the HF/M3 sonar has demonstrated a probability of single-ping detection above 95 percent within the LFA mitigation zone for most marine mammals (see the Navy's Final EIS, Navy 2001, section 2.3.2.2). Combined with the visual monitoring and passive acoustic monitoring protocols, this should minimize the risk of marine mammals being exposed to sound pressure levels in excess of 180 dB.

However, these monitoring protocols can be expected to be more effective with larger animals, like the baleen and toothed whales and adult leatherback and loggerhead turtles. The effectiveness of these monitoring protocols at detecting species like Pacific salmon and the smaller sea turtles remains unknown. Since the HF/M3 sonar is patterned after technology commercial and sports-fishing industries use to locate fish, we will assume that the sonar would locate schooling species like Pacific salmon. If the sonar did not detect schools of Pacific salmon and salmon were exposed to a SURTASS LFA ping, the evidence available suggests that sound pressure levels in excess of 180 dB are likely to have no observable effect on salmon smolt (Knudsen *et al*. 1994 concluded that sound at 150 Hz had no observable effect on Atlantic salmon smolt, *Salmo salar*, at intensities 114 dB above their hearing thresholds of 95 dB).

We would also assume that these monitoring protocols would be less effective at detecting species like olive ridley, Kemp's ridley, smaller leatherback, hawksbill, green, and some loggerhead turtles; the monitoring protocols may not detect some of these species at all.

Although the probability of an interaction between SURTASS LFA sonar and individuals of any of these species is statistically small (the Navy's analyses concluded that the possible number of times a leatherback sea turtle could be in the vicinity of a SURTASS LFA sonar vessel would be less than three out of 18,000 animals per year per vessel; with the monitoring protocols, the Navy concluded that this number would approach zero), the probability could increase depending on the deployment of the SURTASS LFA vessels. The effects of these sound pressure levels on sea turtles remains unknown (Croll *et al*. 1999). Although the decibel levels that could cause potential tissue damage in sea turtles remains unknown, sea turtles are unlikely to be more sensitive to sound-induced tissue damage than marine mammals (Croll *et al*. 1999).

While there is limited empirical evidence (beyond Schlundt *et al*., 2000) that can be used to argue that 180 dB would not injure marine mammals, other observations support this conclusion. For example, toothed whales and some pinnipeds are not highly sensitive to low-frequency sounds (Richardson *et al*. 1995); as a result, they are not likely to be injured or otherwise adversely affected by sound pressure levels in these low frequencies. To gain insights into potential effects of SURTASS LFA sonar on other species, Richardson *et al*. (1995) also extrapolated from human damage risk criteria and suggested that marine mammal auditory systems might be at risk from a single, explosive pulse if the peak received level exceeded 214 to 244 dB (although they emphasized the speculative nature of this extrapolation). Nevertheless, these impulsive thresholds, if they are valid, would suggest that a ping from SURTASS LFA sonar is not likely to injure marine mammal hearing outside of the mitigation zone.

Other studies also support this conclusion. Frankel (1994) estimated the source level for singing humpback whales at between 170 and 175 dB; the average call source level for blue whales was calculated by McDonald *et al*. (2001) to be 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). It seems reasonable to conclude that marine mammals would not be harmed or injured by frequencies at decibel levels that they use in their own vocalizations.

Therefore, unless an animal is within the 180-dB LFA mitigation zone during a ping, animals should not damaged physically or anatomically, including damage to tissues. Because of the mitigation measures, the likelihood of a marine mammal being exposed to these sound pressure levels is very low.

Potential Stranding

Several mass strandings of cetaceans have occurred with some being associated with military operations. Although the majority of these events did not involve listed species and did not implicate SURTASS LFA sonar, we discuss them in this Opinion in response to concerns raised by members of the public that SURTASS LFA sonar could be another cause of marine mammal stranding events. Naval maneuvers have been identified as the potential cause in cetacean strandings in at least two reports (Franzis 1998, Simmonds and Lopez-Jurado 1991). These reports associated offshore naval operations with a mass stranding of Cuvier's beaked whales (*Ziphius cavirostris*) in the eastern Mediterranean Sea in 1996 (Franzis 1998) and mass strandings of Gervais' beaked whales (*Mesoplodon europaeus*), de Blainville's dense-beaked whales (*M. densirostris*), and Cuvier's beaked whales off the coast of the Canary Islands in the late 1980s (Simmonds and Lopez-Jurado 1991).

The North Atlantic Treaty Organization (NATO) Supreme Allied Commander, Atlantic Center Undersea Research Centre that conducted the sonar tests convened panels to review the data associated with the maneuvers in 1996 and beaked whale strandings in the Mediterranean Sea. The report of these panels presented more detailed acoustic data than were available for beaked whales stranded in the Canary Islands (SACLANTCEN 1998). The NATO sonar transmitted two simultaneous signals lasting four seconds and repeating once every minute. The simultaneous signals each were broadcast at source levels of just under 230 dB re 1 μPa at 1 m. One of the signals covered a frequency range from 450-700 Hz and the other one covered 2.8-3.3 kHz. The *Ziphius* strandings in the Kyparissiakos Gulf occurred during the first two sonar runs on each day of 12 and 13 May 1996. The close timing between the onset of sonar transmissions and the first strandings suggests closer synchrony between the onset of the transmissions and the strandings than was presented in Frantzis (1998). However, the Bioacoustics Panel convened by NATO was unable to reach a definitive conclusion due to the lack of evidence of direct physical injury because no viable tissue samples suitable for laboratory analysis were recovered from any of the animals. Their official finding was "An acoustic link can neither be clearly established nor eliminated as a direct or indirect cause for the May 1996 strandings."

Public concern about potential causal relationships between SURTASS LFA sonar and marine mammal stranding resurfaced recently after a beaked whale stranding in the Bahamas in 2000. Fox *et al*. (2001) ruled out natural sound sources as a possible cause of the stranding, which pointed to an anthropogenic source. In 2001, the Joint Interim Report, Bahamas Marine Mammal Stranding Event of 14-16 March 2000 (U.S. Department of Commerce and Secretary of the Navy 2001) concluded that "tactical mid-range frequency sonars onboard U.S. Navy ships that were in use during the sonar exercise in question were the most plausible source of this acoustic or impulse trauma." The report also went on to conclude, "the cause of this stranding event was the confluence of Navy tactical mid-range frequency sonar and the contributory factors acting together." The contributory factors identified included "a complex acoustic environment that included the presence of a strong surface duct, unusual underwater bathymetry, intensive use of multiple sonar over an extended period of time, a constricted channel with limited access, and the presence of beaked whales that appear to be sensitive to the frequencies produced by these sonars." This report also stated that, "SURTASS LFA, another Navy sonar, had no involvement in this event."

This opinion did not re-examine the work of the NATO Bioacoustics Panel or the panel that reviewed the Bahamas stranding; however, our initial reviews of the data available on the strandings lead us to believe that much more information would have to become available before we could assess potential causal relationships between the Navy's operations and the stranding events. Nevertheless, we can evaluate this series of stranding events by applying Mill's method of agreement to infer the probable causal relationship between the beaked whale strandings and the sonars that were being employed in the different naval maneuvers (Mill 1856; see Fox 1991 and Jekel *et al*. 2000 for more recent applications of this method). This method of causal inference evaluates two or more instances of phenomena being investigated (in this instance, the beaked whale stranding) to determine which attributes they have in common; if they only have one common attribute, that attribute is probably causal.

The beaked whale strandings in the Kyparissiakos Gulf and the Bahamas were both associated with naval maneuvers that were using sonars and appeared to precede the stranding events. In the Kyparissiakos Gulf, the naval maneuvers employed both mid-frequency and low-frequency sonars. In the Bahamas, naval maneuvers employed only mid-frequency sonars: the inquiry that followed the beaked whale stranding in the Bahamas concluded that a mid-frequency sonar, of the type used by Navy destroyers, probably caused the deaths of the beaked whales, although other risk factors were also involved (U.S. Dept. Commerce and Secretary of the Navy 2001). The only attribute the two stranding events have in common is the use of mid-frequency sonar, which was considered causal in the Bahamas stranding event. Applying the method of agreement, we would conclude that a mid-frequency sonar was probably causal in the beaked whale strandings in the Kyparissiakos Gulf as well, although, as in the Bahamas, other risk factors may have been operative.

Drawing on information produced by the Scientific Research Program, the sonar associated with LFA has not been known to produce effects that would cause marine mammals to strand. Further, monitoring and mitigation protocols proposed for employment of SURTASS LFA sonar will preclude the sonar's employment in narrow and deep channels surrounded by land such as those in the Bahamas (22-km/12-nm restriction). Frequencies less than 330 Hz would be used and the shut-down criteria for the SURTASS LFA sonar have been expanded to include any detection by the Navy's high-frequency marine mammal monitoring (HF/M3) sonar that is classified as a marine mammal or sea turtle. In most situations, these measures would prevent species from approaching within 2 km (1.1 nm) of the SURTASS LFA source ship during a ping (which would prevent marine mammals from being exposed to sound pressure levels above 180 dB.

Potential Resonance Effects

The concept that resonance will increase stress on tissue to the point of damage is in reality two separate concepts: (1) resonance and (2) tissue damage. As stated by Cudahy and Ellison (2001), resonance does not equal damage and damage is not always linked to resonance. So the issue is not resonance in air/gas cavities, but tissue damage, whether it is caused by resonance or by other means. The potential for *in vivo* tissue damage to marine mammals from exposure to underwater LF sound will occur at a damage threshold on the order of 180 to 190 dB (Cudahy and Ellison, 2002). This includes transluminal (hydraulic) damage to tissues at intensities on the order of 190 dB or greater; vascular damage thresholds from cavitation at intensities in the 240-dB regime; tissue shear damage at intensities on the order of 190 dB or greater; and tissue damage in air-filled spaces at intensities above 180 dB.

Another potential mechanism for tissue damage could be “bubble growth” (similar to the “bends” in divers) in supersaturated tissues. Crum and Mao (1996) hypothesized that the received level would have to exceed 190 dB in order for there to be the possibility of significant bubble growth due to supersaturation of gases in the blood.

An analysis by Cudahy and Ellison (2001) of the potential for resonance from LFA signals to cause injury supports the conclusion 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. While limited empirical evidence (beyond Schlundt *et al*., 2000) supports 180 dB as level that is "safe" for marine mammals, information on marine mammal vocalizations suggest that 180 dB is not likely to injure marine mammals. 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). Since it is unlikely that whales would vocalize at sound pressure levels that would injure or produce resonance effects in other members of their species, it seems reasonable to infer that 180 dB is not likely to injure them.

Therefore, unless an animal is in the 180-dB LFA mitigation zone, NMFS believes that present scientific information indicates that there should be no physical damage to marine mammal body systems or tissues at sound pressure levels less than 180 dB. Because of the mitigation measures, the potential taking of a marine mammal within the 180-dB mitigation zone is considered minimal. If marine mammals vocalize at these levels, it is realistic to believe that these species have also evolved mechanisms to protect themselves and conspecifics from high sound pressure level vocalizations.

Potential Behavioral Effects

Based on the evidence available, SURTASS LFA sonar operations as defined are not likely to kill or injure marine mammals, including threatened or endangered species; however, little is known about the effect of short-term disruptions of a marine mammal's normal behavior (Richardson *et al*. 1995). Most of the evidence available suggests that most sources of disturbance do not directly kill or injure marine mammals. The evidence available also does not lead us to expect stranding or resonance effects from the SURTASS LFA signal. The only category of effects that remains is behavioral effects, which can consist of behavioral changes that have either short-term or long-term consequences for populations of threatened and endangered species. Existing studies of behavioral effects of low-frequency, man-made sounds in marine environments are inconclusive, partly because of their limited ability to detect behavioral changes that are significant to the biology of the individuals that are being observed. Those studies are further complicated by the variety of responses that can occur within a single species of marine mammals, which can exhibit a wide range of responses to man-made noise that can vary by individuals and their circumstances. Under some circumstances, some individuals will continue the normal activities in the presence of high levels of man-made noise; in other circumstances, other individuals may avoid the source at much lower received levels (Richardson *et al*. 1995).

Data generated by the research program showed that some whales responded to SURTASS LFA sonar signals by either changing their level of vocal activity, moving away from or approaching the SURTASS LFA source vessel, or both. The Scientific Research Program showed that there were statistically significant avoidance responses from gray whales migrating along the coast when the sound source was inshore 2 km from the coast), but not offshore (4 km from the coast; Buck and Tyack 2000) and the level of response was proportional to the level of sound received by the whale. The research results also showed that some humpback whales reduced their vocal activity. Those that continued singing, increased song length, but the tendency for these responses did not increase with increasing received levels (Clark and Tyack 1998; Miller *et al*. 2000). Nevertheless, in all cases, responding whales resumed normal activities within a few tens of minutes after initial exposure to LFA signals. The data collected by the research program demonstrated that some portion of the whales exposed to the SURTASS LFA sonar exhibited behavioral responses, but those responses were short-lived.

Scientists have also been concerned that prolonged exposure to 120 dB might cause temporary or permanent threshold shifts in toothed whales at their most sensitive frequency (Richardson *et al*. 1993). Temporary threshold shift (TTS) is a change in the threshold of hearing (a change in the quietest sound an animal can hear) that could temporarily affect an animal's ability to hear calls, echolocation sounds, and other ambient sounds. As such, although these shifts are not considered injurious to a marine mammal, they could temporarily disrupt an animal's behavioral patterns (which would meet the MMPA definition of Level B harassment). The best information available indicates that the distortion and dysfunction of sensory tissue observed during TTS are only temporary and are fully reversed upon recovery (i.e., occasional TTS produces no permanent tissue damage to the ear, only the temporary nondestructive impairment of tissue that fully recovers). Permanent threshold shift is a deterioration of hearing due to prolonged or repeated exposure to sounds that accelerate the normal process of gradual hearing loss (Kryter 1985), and the permanent hearing damage due to brief exposure to extremely high sound levels (Richardson *et al*. 1995).

Recent research has reduced concerns about the potential relationship between SURTASS LFA sonar and threshold shifts in marine mammals, although the research has not eliminated those concerns. For example, Schlundt *et al*. (2000) showed that bottlenose dolphins experience onset of masked TTS (defined as 6 dB of shift) from a one-second, 3-75 kHz, exposure at received levels of approximately 192 dB. Assuming a 3 dB exchange rate (e.g., the same amount of shift would result from reducing the intensity by 3 dB and doubling the exposure time (Finneran *et al*., 2000), these odontocetes could experience TTS from a 16-second exposure to a 180-dB sound at their best frequency, a 32-second exposure at 177 dB, etc. Since these are estimates for marine mammals that are mid-frequency specialists responding to mid-frequency sound levels, NMFS preliminarily believes that low frequency marine mammal specialists would incur TTS at similar levels and duration in response to low frequency sounds.

The specific responses of whales to LFA transmissions have been the subject of separate studies by Bowles *et al*. (1994), Croll *et al*. (2001), Miller *et al*. (2000), and U.S. Navy (2000b) and has been associated with cetacean strandings in at least two other reports (Franzis 1998, Simmonds and Lopez-Jurado 1991). The studies identified increases in the length of whale songs (Miller *et al*. 2000) and changes in respiration and reorientation rates (Bowles *et al*. 1994). Although Croll *et al*. (2001) did not report whales responses that were statistically or biologically significant, they expressed concern about the long-term, cumulative effects of anthropogenic, low-frequency noise on marine mammals.

There are no published data on marine mammals' responses to repeated exposure to intermittent, low frequency sound of the type that would be generated by SURTASS LFA. Prior to the

Research Program, the best information regarding whale responses to continuous, low frequency, anthropogenic noise was summarized by Richardson *et al*. (1995b), who concluded that some marine mammals would tolerate continuous sound at received levels above 120 dB re 1 _Pa for a few hours. However, other marine mammals avoided the source when sound levels reached ~120 dB. It is doubtful that many marine mammals would remain for long in areas where received levels of continuous underwater noise are >140 dB, at frequencies to which the animals are most sensitive. Richardson *et al*. (1995a) and Richardson (1997, 1998) also reported on controlled playback experiments and observations around actual industrial sources that showed bowhead whales avoided drill ship noise at estimated received levels of 110 to 115 dB and seismic sources at estimated received levels of 110 to 132 dB. These reactions were not severe and the animals habituated themselves to the sound. It should also be noted that the Arctic Sea (where SURTASS LFA sonar will not operate) is a pristine area with low levels of industrial and shipping noise.

These conclusions are supported by other studies of whale responses to low frequency, anthropogenic sounds. For example, bowhead (*Balaena mysticetus*) and gray whales (*Eschrichtius robustus*) clearly showed patterns of short-term, behavioral disturbance in response to a variety of actual and simulated vessel activity and noise (Richardson *et al.* 1985; Malme *et al.* 1983). Humpback whales showed similar patterns on their summering grounds (Baker and Herman 1987) and on their wintering grounds (Bauer 1986) in response to vessel noise.

Another study examined responses of gray whales migrating along the California coast to various sound sources located in their migration corridor (Malme *et al*. 1983, 1984). Gray whales 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. This study was replicated in Phase II of the Scientific Research Program using SURTASS LFA sonar stimuli. However, the Phase II research demonstrated that it may be invalid to apply the inshore (2 km from shore) response model (when 50 percent of the whales avoided SURTASS LFA sonar stimuli at received levels of 141 ±3 dB) to sources that are offshore (4 km from shore) of migrating whales, and that whales did not avoid offshore sources at received levels of 140 dB.. This implies that the inshore avoidance model – in which 50 percent of the whales avoid exposure to levels of 141 ±3 dB – may not be valid for whales in proximity to an offshore source (Buck and Tyack 2000).

The third phase of the research program examined potential effects of SURTASS LFA sonar transmissions on singing humpback whales. These whales showed some apparent avoidance responses and cessation of song during specific LFA sound transmissions at received levels ranging from 120 to 150 dB. However, an equal number of singing whales exposed to the same levels showed no cessation of song during the same LFA sound transmissions. Of the whales that did stop singing, there was little response to subsequent LFA sound transmissions; most joined with other whales or resumed singing within less then an hour of the possible response. Those that did not stop singing, sang longer songs during the period of LFA transmissions, and returned to baseline after transmissions stopped (Miller *et al*., 2000). Further analysis is required to establish how often male humpbacks stop singing in the absence of the SURTASS LFA sonar transmissions, and to evaluate the significance of the song cessation observed during playbacks.

Until additional data become available, the weight of available evidence suggests that SURTASS LFA sonar will elicit responses that, at worst, can be characterized as "disturbance" that would be expected to have only minimal effects on marine mammal populations (see also Clark and Fistrup 2001). Nevertheless, recovery plans for the endangered whales identified disturbance as one of the principal human-related factors impeding the recovery of these whale species (NMFS 1991a, 1991b).

Richardson *et al*. (1995) noted that avoidance reactions are the most obvious manifestations of disturbance in marine mammals. Migrating whales may avoid stationary sources on the original course by deflecting their course slightly as they approach the source (LGL and Greenridge 1987 in Richardson *et al*. 1995). In cetaceans, mean durations of surfacings and dives, number of blows per surfacing, and intervals between successive blows often are affected by man-made noise. When bowhead and gray whales are exposed to various types of man-made noise, their surfacings become unusually brief with fewer blows per surfacing (Richardson *et al*. 1985, 1986, 1990; Malme *et al*. 1988; Richardson and Malme 1993). These changes in surfacing, respiration, and diving behavior may be indicators of stress, although their consequences on the population ecology of the animals that are affected remain unknown.

Taken together, the data generated during the three phases of the LFS Scientific Research Program did not support the initial hypothesis that most baleen whales exposed to received levels near 140 dB would exhibit behavioral changes and avoid the area. These experiments, which exposed baleen whales to received levels ranging from 120 to about 155 dB, detected only minor, short-term behavioral responses whose potential effects on behavior that is important to the population ecology of these species remains unknown. The Scientific Research Program also examined short-term behaviors, such as feeding, that relate to birth and growth rates, which would affect population ecology.

These conclusions are supported by other studies of whale responses to low frequency, anthropogenic sounds. For example, bowhead (*Balaena mysticetus*) and gray whales (*Eschrichtius robustus*), showed clear patterns of short-term, behavioral disturbance in response to a variety of actual and simulated vessel activity and noise (Richardson *et al.* 1985; Malme *et al.* 1983). Richardson *et al*. (1995) noted that whales have been seen within a few kilometers of operating seismic vessels, although they added that any discomfort the seismic sound pulses may have caused remains unknown. Humpback whales showed similar patterns on their summering grounds (Baker and Herman 1987) and on their wintering grounds (Bauer 1986) in response to vessel noise. Richardson *et al*. (1995) argued that intermittent pulses with peak levels between 160 to 180 dB are less likely to cause discomfort than continuous sounds at the same sound pressure levels.

As noted below (see *Potential Masking* subsection), Watkins (1986) review of data on the reactions of fin, humpback, right and minke whales that were exposed to continuous, broadband low-frequency shipping and industrial noise in Cape Cod Bay is informative. Watkins (1986) concluded that underwater sound was the primary cause of a reaction in these species of whales and that whales responded to acoustic stimuli in their range of hearing. Watkins also noted that whales showed the strongest 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, Watkins (1986) 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 have generally habituated to the continuous, broad-band, noise of Cape Cod Bay while right whales did not appear to change their response.

This study covered a long enough period (the data covered 25 years of time) to provide some insight into possible long-term effects of low-frequency noise on whales, particularly since the four whale species would be exposed to continuous, low-frequency noise from shipping and other industrial sources. Given that whales in Cape Cod Bay reacted negatively to these continuous sources of anthropogenic sounds only under specific circumstances and, over time, habituated to these sounds (rather than abandon the area), it seems unlikely that an additional, intermittent signal lasting from 6 to 100 seconds that is designed to mimic background, low frequency sound would have a greater, negative effect on, at least, these species whales (fin, humpback, right, and minke whales). The studies associated with the LFS Scientific Research Program suggest the same conclusions may also apply to blue, gray, and fin whales.

In conclusion, based on the data available, NMFS believes the proposed action is likely to produce short-term changes in the behavior of differing percentages of specific whale populations (see Table 4). Once NMFS reviews the Navy's annual requests for letters of authorizations, we will examine the specific operating areas to assess potential adverse effects on specific populations of different marine mammal species. However, NMFS believes the best scientific and commercial data available, although limited, would not lead to a conclusion that these short-term effects would have long-term, adverse consequences for the biology or ecology of the individual whales exposed to the LFA signal. More importantly, the best scientific and commercial data available suggests that the effects of the proposed SURTASS LFA sonar may have short-term, adverse effects on individual whales, but is not likely to adversely affect the population dynamics of endangered whales in ways that would reduce their reproduction, numbers, or distribution.

Potential 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 (Erbe and Farmer 2000, Tyack 2000). Masking these acoustic signals can disturb the behavior of individual animals, groups of animals, or entire populations. Richardson *et al*. (1995b) 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 (which could include low frequency sound like SURTASS LFA sonar) is most likely to affect some species' ability to detect communication calls and natural sounds (i.e., surf noise, prey noise, etc.; Richardson *et al*., (1995b)).

Baleen whales

Like Richardson *et al*. (1995), we assume that baleen whales are very sensitive to low-frequency sounds. As a result, masking effects could be significant for many of these whales because they vocalize at low frequencies and are thought to have hearing that is sensitive at the SURTASS LFA sonar frequencies. This is especially true for those animals that use the same frequency bands as SURTASS LFA sonar. For example, Dahlheim *et al*. (1984) concluded that gray whales in the San Ignacio Lagoon, Baja, California shifted the frequencies of their vocalizations away from the predominant ambient noise producers in the lagoon to overcome masking effects.

In contrast, Biassoni *et al.* (2001) concluded that the intermittent sounds produced by LFA sonar were unlikely to mask humpback whale songs, but the similarities of its sounds to those produced by the whales could cause some humpback whales to perceive LFA sonar as a competing male. Based on their studies, they concluded that humpback whales could adapt to the presence of LFA sonar and concluded that singing whales could compensate for interference from sound sources like LFA sonar.

As an issue for marine mammals, continuous, broadband low-frequency shipping and industrial noises are likely to pose a greater threat to marine mammals than SURTASS LFA sonar sound because a continuous sound, such as noise from a ship, cannot be masked by an intermittent LFA transmission. A review of the reactions of fin, humpback, right and minke whales that were exposed to continuous, broadband low-frequency shipping and industrial noise in Cape Cod Bay is informative. Watkins (1986) concluded that underwater sound was the primary cause of a reaction in these species of whales and that whales responded to acoustic stimuli in their range of hearing. Nevertheless, he 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.

Although these data were limited in time (they only covered 25 years of time, which may represent only one generation for some species of cetaceans) and spatial extent and only focused on four species of whale, they represent data from an area where whales would have been expected to exhibit somewhat long-term effects of masking from continuous shipping and industrial noise. Given that whales in Cape Cod Bay reacted negatively to these continuous sources of anthropogenic sounds only under specific circumstances and, over time, habituated to these sounds (rather than abandon the area), it seems unlikely that masking from an intermittent source like SURTASS LFA sonar would appreciably affect these species of whales.

Despite these studies, our knowledge of the function of most marine mammal calls is limited, so it would be difficult to reach definitive conclusions about the effects of reducing detection distances on the health or viability of marine mammal populations. Making inferences from the available evidence is further complicated because some marine mammals can apparently compensate for masking (Richardson *et al*. 1995), although the compensatory mechanisms, their effectiveness, and the costs to the compensative individuals (in terms of energy expenditures or stress) remain unknown. Nevertheless, the available evidence does not lead us to expect masking to cause reductions in the reproduction, numbers, or distribution of threatened or endangered mysticete whales.

Toothed whales

The echolocation calls of toothed whales are subject to masking by high frequency sound. 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). Since echolocation calls occur at much higher frequencies than SURTASS LFA sonar, the extent of upward masking (i.e., low frequencies masking high frequencies) would be limited. There is also evidence that the directional hearing abilities of odontocetes are useful in reducing masking at the high frequencies used for echolocation, but not at the low-moderate frequencies used for communication (Zaitseva *et al*. 1980).

Although low frequency hearing has not been studied in many odontocete species, those species that have been tested (beluga, killer whale, false killer whale, Risso's dolphin, and bottlenose dolphin) exhibit poor audiometric and behavioral sensitivity to low frequency sound. For sounds dominated by low frequency components, the maximum radius of audibility for most odontocete species may often be noise-limited when sensitivity is good, and sensitivity-limited when sensitivity is poor. At a maximum 20 percent duty cycle, it is anticipated that any masking of odontocetes would be temporary (i.e., at least 80 percent of the time an animal would be able to perceive incoming signals through low frequency sounds). The possibility of effective masking would only occur for environmental sounds that happen during the ping transmission (maximum 100 seconds) and are at, or at least close to, the frequencies in the 30-Hz-wide bandwidth signal, during the 10 seconds the SURTASS LFA sonar was transmitting in that bandwidth. As a result, the available evidence does not lead us to expect masking to directly reduce the reproduction, numbers, or distribution of threatened or endangered odontocetes or elicit behavioral responses that would reduce the reproduction, numbers, or distribution of these species.

Interactions Between SURTASS LFA Sonar and Background Sound Levels

Several investigators and organizations have expressed concern about the "cumulative impact" (in the NEPA sense of the term) of marine sounds on the ocean environment and its organisms (NRDC 1994, 2001, Richardson *et al*. 1995). Any man-made sound that is strong enough to be audible (detectable above natural background noise) will increase total background levels and could interfere with an animal's ability to detect sound signals if the signal is weak relative to total noise levels. Concern about the cumulative impact of man-made sounds focuses on impacts from individual actions that are insignificant or minor when considered in isolation, but combine to produce effects that are greater than any individual action (either because the effects are synergistic - effects that occur when two or more phenomena interact - multiplicative, or additive). In this Opinion, our principal concern is the effect of adding SURTASS LFA sonar to underwater ambient noise levels.

SURTASS LFA sonar operations will add sound to ambient oceanic noise levels, which, in turn, could have cumulative impacts on the ocean environment, including listed species. When transmitting, SURTASS LFA sonar will add to regional noise levels. Unfortunately, there are no reliable methods for assessing these potential cumulative impacts. The U.S. Navy conducted computer simulations to assess the potential cumulative impacts of SURTASS LFA sonar (Navy 2001; section 4.4.1 through 4.4.4). That assessment concluded that the "cumulative impacts" of SURTASS LFA sonar would be "extremely small" because SURTASS LFA would be operated for a relatively brief period of time on an annual basis (estimated maximum of 432 hours per vessel per year which equates to 18 days per system or 36 days for the two systems being considered in this Opinion); the system would operate at a low duty cycle (on no more than 20 percent of the time), and for relatively short periods of time in any given area; the system would not be stationary; a maximum of only four systems would be operational (with usually only 1-2 at sea at any one time, and almost always in separate oceans); and all observations made during the Scientific Research Program suggest that impacts terminate when transmissions stop.

Although SURTASS LFA sonar would add very small amounts of energy to the world's ocean environment, NMFS remains concerned about the potential cumulative impacts of these sound sources on the oceans and the biota that inhabit them. For example, underwater noise associated with extensive vessel traffic has been documented to have caused gray whales to abandon some of their habitat in California for several years (Gard 1974, Reeves 1977). Salden (1988) suggested that humpback whales avoid some nearshore waters in Hawaii for the same reason.

Richardson *et al*. (1995) provided extensive information and arguments about the potential cumulative effects of man-made noise on marine mammals. Those effects included masking, physiological effects and stress, habituation, and sensitization. Those concerns were echoed by Clark and Firstrup (2001), Michel *et al.* (2001), NRDC (2001), and others. Although all of these responses have been measured in terrestrial animals reacting to airborne, man-made noises, those studies are counterbalanced by studies of other terrestrial mammals that did not exhibit these responses to similar acoustic stimuli.

Richardson *et al*. (1995) also recommended several operational measures to minimize the effects of man-made sounds on marine mammals. These included minimizing source levels, minimizing duty cycles, and gradually increasing projected sound levels to allow animals to move away from the source before source levels peak. The Navy has clearly included these mitigative measures into the operations SURTASS LFA . Nevertheless, these measures will minimize, but not eliminate, the potential cumulative impacts of SURTASS LFA sonar; those impacts will probably remain unknown.

Effects of the HF/M3 Sonar

The source level required for the HF/M3 sonar to effectively detect marine mammals (and possibly sea turtles) out to the 180-dB LFA mitigation zone under the most adverse oceanographic conditions (low echo return and high ambient noise) is on the order of 220 dB. The Navy designed the HF/M3 sonar to be as benign as possible within the marine environment in order to minimize potential effects to marine mammals and sea turtles. These features include:

1. The HF/M3 sonar source frequency is >30 kHz, which pushes its frequency band well away from the best hearing bandwidth of mysticetes, pinnipeds, and sea turtles, but within the best hearing bandwidth of odontocetes;

2. A duty cycle that is variable, but below 10 percent;

3. A maximum HF/M3 sonar pulse with a duration of 40 milliseconds (msec). Ridgway *et al*. (1997) and Schlundt *et al*. (2000) reported that measured temporary threshold shifts in bottlenose dolphins for a 20 kHz, 1-second pulse occurred at response levels of 193-196 dB. For a 30 kHz, 40-msec pulse, the estimated range from the HF/M3 sonar of 193 dB response level would be 22 m (72 ft); and

4. A transmission loss that is very high because of the high frequency of the sound source.

In addition, as supplementary safety measures, the following operational procedures would be applied to operation of the HF/M3 sonar:

1. The HF/M3 sonar source level would be ramped up over a five-minute period to alert a marine mammal that was close to the sonar and provide it time to move away from the sound source; and

2. The HF/M3 sonar source level would be reduced if a marine mammal or sea turtle was detected approaching the sonar. As an animal approaches the sound source, the source level would be adjusted to ensure the received level at the animal remains below 180 dB.

The application of these operational procedures reduces potential impacts of the HF/M3 sonar on marine mammals and sea turtles to negligible levels. The HF/M3 sonar is basically a “fish finder” sonar and the evidence available suggests that it does not harm fish.

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 in this biological 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.

NMFS expects commercial and recreational fisheries managed by coastal state to continue within the action area for the foreseeable future. NMFS also expects whale watching operations and aircraft tours to continue for the foreseeable future. NMFS believes it would be virtually impossible to measure or detect the degree to which these activities may disturb or harass listed whales within the action area. However, based on the increasing size of these whale populations and their increased occurrence in these waters, NMFS concludes that these activities do not appear to have long-term, adverse effects on these whale populations.

More importantly for these endangered whales, NMFS expects commercial vessels to continue to transit the action area. For example, the ports of Jacksonville and Port Everglades, Florida; Baltimore, Maryland; Wilmington, Delaware; Philadelphia, Pennsylvania; New York, New York; and Boston, Massachusetts support some of the country's strongest maritime economies. About 17 million tons of waterborne cargo pass through the Port of Jacksonville, Florida which receives about 1,600 vessels each year moving between the U.S. and South America, Europe, and the Caribbean. About 4.8 million tons (short tons) pass through the Port of Wilmington, Delaware, which receives about 400 vessels each year. About 56 million tons of waterborne cargo passed through the Port of New York in 1998. About 1.3 million tons of general cargo, 1.5 million tons of bulk cargo, and 12.8 million tons of bulk fuel cargo pass through the Port of Boston, Massachusetts which receives more than 62 ship calls, 350 container vessels, and 1,700 bulk cargo vessels each year. In addition, about 60 cruise vessels sail from the Port of Boston each year[6]. Analyses of the U.S. west coast ports would yield the similar results.

NMFS expects commercial traffic into and out of ports adjacent to the action area to continue into the foreseeable future. The best scientific and commercial data available provides no specific information on the risk this level of commercial traffic poses to listed marine animals in the action area, but we would expect this level of commercial traffic to pose a risk of ship strikes that would continue to result in observed death or serious injury of whales in numbers similar to those observed between 1994 and 1999 (1 blue whale, 1 sei whale, 2 fin whales, 6 right whales).

INTEGRATION AND SYNTHESIS OF EFFECTS

The following discussions summarize the probable risk SURTASS LFA sonar transmissions 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 the response analyses that were also presented previously.

Steller sea lions. About 10 percent of the Steller sea lion population in the eastern Pacific Ocean have a chance of being exposed to SURTASS LFA transmissions. Because of the restrictions that prevent the SURTASS system from being operating within the 200-meter isobath of the United States, few of the sea lions in the Gulf of Alaska, eastern Alaskan peninsula, Kodiak Island area, and the eastern population are likely to be exposed to these transmissions. Animals occupying rookeries and haulouts in the Bering Sea and along the northern coast of the Alaskan peninsula, which includes some of the largest rookeries in the western population, are not likely to be exposed. Sea lions on rookeries and haulouts in the Aleutian Island chain, which currently support the smallest and most vulnerable proportion of the western population, have the highest risk of exposure to SURTASS LFA transmissions. Based on the Navy's models, these animals would receive levels ranging from slightly less than 160 dB to less than 120 dB with a unimodal distribution centered midway between 150 and 145 dB tapering quickly to 130 dB.

Steller sea lions generally make shallow dives (Kenyon 1952, NMFS unpublished information), which would limit their exposure to SURTASS LFA sonar transmissions. However, Steller sea

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[6] These data were derived from information available the internet websites for each of the ports named.

lions will dive to 277 meters while foraging, depths that could expose them to SURTASS LFA signals. Regardless, a significant number (10 percent) of Steller sea lions have a probability of being exposed to SURTASS LFA transmissions. Sea lions appear to vocalize as part of their social behavior and are able to hear well in and out of water; however, there are no data on the response of sea lions to low frequency sounds. However, data from studies of the effects of low frequency sounds on elephant seals (*Mirounga* spp.), which are considered more sensitive to low frequency sounds than other pinnipeds (Croll *et al.* 1999, Kastak 1996, LeBoeuf and Peterson 1969), suggest that elephant seals did not experience short-term changes in behavior in response to low frequency sounds. Based on the Steller sea lion's limited sensitivity to low frequency sound (Croll *et al*. 1999, Richardson 1995), we would not expect received levels of SURTASS LFA transmissions to reduce the reproduction, numbers, or distribution of this sea lion; as a result, these transmissions would not be expected to appreciably reduce this sea lions likelihood of surviving and recovering in the wild.

Guadalupe fur seals. These fur seals could be exposed to SURTASS LFA transmissions throughout their distribution from Guadalupe Island north to the Channel Islands. Based on the results of the Navy's models, about 0.45 percent of these fur seals have a chance of being exposed to SURTASS LFA transmissions at received levels less than 180 dB (these data are from simulations for Site 8, which represents the waters surrounding San Nicholas Island, California). Of the animals that are exposed to these transmissions, received levels would range between 120 dB (which is generally considered ambient in marine ecosystems) and 160 dB.

Like sea lions, fur seals generally have shallow dives (Gallo-Reynoso 1994, Reeves *et al*. 1992), which would limit their exposure to SURTASS LFA sonar transmissions. Fur seals also vocalize as part of their social behavior and are able to hear well in and out of water; however, there are no data on the responses of fur seals to low frequency sounds. The data we cited from studies of the effects of low frequency sounds on elephant seals are probably indicative of the sensitivity of fur seals as well and suggest that Guadalupe fur seals have limited sensitivity to low frequency sound (Croll *et al*. 1999, Richardson 1995). We would not expect received levels of SURTASS LFA transmissions to reduce the reproduction, numbers, or distribution of this fur seal; as a result, these transmissions would not be expected to appreciably reduce this fur seal's likelihood of surviving and recovering in the wild.

Hawaiian monk seals. Although Hawaiian monk seals generally reside in coastal waters near haul-out areas forage in deep water and dive to at least 490 m (1,608 ft; Reeves *et al*. 1992), which could expose them to low frequency sounds from SURTASS LFA. However, the Navy's simulation models suggest that Hawaiian monk seals would be exposed to SURTASS LFA transmissions northwest of the Island of Kauai, where 2.39 percent of the population might be exposed to SURTASS LFA transmissions (these data are from simulations for Sites 6, 7, and 12, which represent the waters surrounding the Islands of Kauai and Oahu in Hawaii). Nevertheless, for those monk seals that would be exposed to these transmissions, received levels would range from 130 dB to 160 dB.

Hawaiian monk seals have their most sensitive hearing at 12 to 28 kHz; their high frequency sensitivity drops off sharply above 30 kHz (Thomas *et al*. 1990b). Below 8 kHz, Hawaiian monk seals have less sensitive hearing than other pinnipeds. Based on the monk seal's limited sensitivity to low frequency sound (Croll *et al*. 1999, Richardson 1995), we would not expect received levels of SURTASS LFA transmissions to reduce the reproduction, numbers, or distribution of this monk seal; as a result, these transmissions would not be expected to appreciably reduce this monk seal's likelihood of surviving and recovering in the wild.

Baleen Whales

As discussed previously, endangered whales have the greatest probability of being detected by the monitoring systems associated with the SURTASS LFA sonar systems, which gives them the lowest probability of being exposed to signals at received levels approximating 180 dB.

We assume that baleen whales are very sensitive to low-frequency sounds. Despite the limited number of studies, the available evidence suggests that the risk of injury, masking, stranding, resonance effects, or behavioral effects in blue whales and other baleen whales is very low. For example, the information available on bowhead whales, which have very sensitive hearing and are extremely sensitive to noise, suggests that bowhead whales will alter their migratory pathways to avoid industrial sound sources and may reduce their calling rates (Richardson *et al.* 1995), although these reactions varied by season and ambient sound levels. Beyond these short-term avoidance reactions, these studies provide no evidence of responses that might imply reduced health in individual whales that were exposed to industrial sound sources.

The best scientific and commercial data available suggest that SURTASS LFA sonar transmissions could elicit short-term effects on baleen whales that are not known to have long-term, adverse consequences for the biology or ecology of the individual whales exposed to the LFA signal. More importantly, the best scientific and commercial data available suggest that SURTASS LFA sonar may have short-term, adverse effects on individual whales, but is not likely to adversely affect the population dynamics of endangered whales in ways that would reduce their reproduction, numbers, or distribution. As a result, these transmissions would not be expected to appreciably reduce these baleen whales' likelihood of surviving and recovering in the wild.

Blue whales. Based on the results of the Navy's models, blue whales had the second highest probability of being exposed to SURTASS LFA sonar (Table 4): about 16 percent of the blue whale population in the North Atlantic Ocean and 6 to 8 percent in the North Pacific Ocean could be exposed to SURTASS LFA sonar depending upon how SURTASS LFA vessels were deployed. The highest proportions of blue whales would be exposed off northern Europe, particularly in the Denmark Strait (Site 17) and off Bergen, Norway (Site 18), which were responsible for 13.77 and 9.83 percent of the exposure, respectively (Note: this analysis was based on the four vessels that were projected to be deployed instead of the 2 vessels that will actually be deployed). The lowest proportions of the exposure were off the coasts of South America. Blue whales in the Southern Ocean (the Antarctic Convergence Zone, which is an offshore biologically important area) are not likely to be exposed to LFA signals at any detectable level unless these animals migrate out of the Southern Ocean.

The intensities of exposure would vary with geographic location. In the Denmark Straits and Bergen, Norway regions, blue whales would be exposed to received levels ranging from about

120 dB to about 170 dB, with the highest proportion exposed to received levels between 150 and 160 dB. In the north Gulf of Alaska, for example, blue whales could be exposed to received levels ranging from about 130 dB to 165 dB, with a unimodal distribution centered between 145 and 150 dB.

Bowhead whales. Because of the operational restrictions associated with SURTASS LFA, the only bowhead whales that are at risk of being exposed to SURTASS LFA transmissions are the Sea of Okhotsk and Spitsbergen populations. Although the Navy did not model potential exposure of bowhead whales, we used the exposure risk of the western Pacific gray whales, about 5 percent, as an index of the proportion of the Sea of Okhostk bowhead whale population that could be exposed to SURTASS LFA transmissions. Similarly, we have no specific information on received levels for bowhead whales, but received levels for gray whales ranged from slightly less than 150 dB to less than 120 dB with a unimodal distribution peaking at slightly less than 145 dB, quickly tapering to 130 dB.

Gray whales. About 5 percent of the western Pacific gray whale population has a chance of being exposed to SURTASS LFA transmissions depending upon how the vessels are deployed. Based on the results of the Navy's models, the western Pacific population of gray whales had the highest probability of being exposed to SURTASS LFA sonar south of Japan (Table 4). The coastal migratory habit of gray whales would protect them from being exposed to sound pressure levels greater than 180 dB. In the north Pacific Ocean, received levels for gray whales (model runs for the eastern Pacific population of gray whales, which are not listed) ranged from slightly less than 150 dB to less than 120 dB with a unimodal distribution peaking at slightly less than 145 dB, quickly tapering to 130 dB.

Fin whales. Based on the results of the Navy's models, fin whales had the highest risk of being exposed to SURTASS LFA sonar in the Straits of Sicily, in the Mediterranean Sea, where about 8 percent of the fin whale population could be exposed to SURTASS LFA transmissions (Table 4). Elsewhere in the Atlantic and Pacific Oceans, about 1 percent of the fin whales had a risk of being exposed to SURTASS LFA transmissions, although the percentage was considerably less than 1.0 at most sites. In the Atlantic Ocean, received levels for fin whales ranged from slightly less than 170 dB to less than 120 dB with a bimodal distribution centered between 135 and 140 dB, quickly tapering to 170 dB.

Humpback whales. Based on the results of the Navy's models, humpback whales had the highest risk of being exposed to SURTASS LFA sonar in the Gulf of Alaska, where about 12 percent of the population could be exposed to SURTASS LFA transmissions. Humpback whales faced their second highest risk of exposure to those transmissions in Denmark Strait region, where 5.33 percent of the population could be exposed to SURTASS LFA transmissions. Elsewhere in the Pacific and eastern North Atlantic Ocean, about 3 to 4 percent of the humpback whales had a chance of being exposed to SURTASS LFA transmissions. In the north Pacific Ocean, received levels for humpback whales ranged from slightly less than 160 dB to less than 120 dB with a unimodal distribution centered between 145 and 150 dB, quickly tapering to 130 dB.

Right whales. About 4 percent of the right whales in the eastern north Pacific and about 3 percent of the right whales in the western north Atlantic Ocean have a chance of being exposed to

SURTASS LFA transmissions (again depending on how the vessels are deployed). Because of the coastal habit of right whales, the restrictions that limit SURTASS LFA transmissions to less than 180 dB within the 200-meter isobath, and transmission losses, few of the right whales along the Atlantic coast are likely to be exposed to received levels above ambient. Received levels ranged from about 170 dB to less than 120 dB with the distribution peaking between 125 and 130 dB and tapering off at a regular rate to 170 dB.

Although the specific distribution of Pacific right whales is largely unknown, these animals are believed to have the same affiliation with coastal areas as right whales in the Atlantic; however, Pacific right whale on foraging grounds off Bristol Bay in the eastern Bering Sea are not likely to be exposed to SURTASS LFA transmissions. Less than 2 percent of the right whales in the southern Pacific had a chance of being exposed to these transmissions. Their received levels ranged from about 148 dB to less than 120 dB with a unimodal distribution centered at slightly less than 145 dB (which quickly tapered off to 130 dB).

Sei whales. Based on the results of the Navy's models, sei whales had the highest risk of being exposed to SURTASS LFA sonar off Sable Island Banks (east of Nova Scotia), where about 3.25 percent of the population could be exposed to SURTASS LFA transmissions. Humpback whales had their second highest risk of exposure to those transmissions in Denmark Strait region and Northeast Norwegian Basin, where 2.1 and 2.5 percent of the population, respectively, could be exposed to SURTASS LFA transmissions. Elsewhere in the Atlantic Ocean, about 4 to 5 percent, and less than 1 percent of the sei whales in the Pacific Ocean had a chance of being exposed to SURTASS LFA transmissions. Received levels for sei whales in the western north Atlantic ranged from slightly more than 170 dB to less than 120 dB with a distribution peaking between 125 and 140 dB.

Sperm whales. Based on the results of the Navy's models, sperm whales had the highest risk of being exposed to SURTASS LFA sonar in the Mediterranean Sea, particularly the Straits of Sicily and Leventine Sea, where about 5 and 3 percent of the population, respectively, could be exposed to SURTASS LFA transmissions. Elsewhere in the Pacific and eastern North Atlantic Ocean, less than 1.0 percent of the sperm whales had a chance of being exposed to SURTASS LFA transmissions. Received levels for sperm whales ranged from slightly more than 160 dB to about 120 dB in the both regions, although a larger percentage of sperm whales in the Straits of Sicily would be exposed to higher received levels.

If exposed to SURTASS LFA sonar transmissions, the evidence available suggests that sperm whales, like other toothed whales, are not very sensitive to low-frequency sounds. Despite the limited number of studies, the available evidence suggests that the risk of injury, masking, stranding, resonance effects, or behavioral effects in sperm whales is very low. The best scientific and commercial data available suggests that SURTASS LFA sonar transmissions is not likely to elicit short-term effects on sperm whales that are not known to have long-term, adverse consequences for the biology or ecology of the individual whales exposed to the LFA signal. Therefore SURTASS LFA sonar is not likely to have short-term, adverse effects on individual whales and is not likely to adversely affect the population dynamics of sperm whales in ways that would reduce their reproduction, numbers, or distribution. As a result, these transmissions would not be expected to appreciably reduce the sperm whales' likelihood of surviving and recovering in the wild.

Sea Turtles (Green sea turtle, Hawksbill sea turtle, Kemp's ridley sea turtle, Leatherback sea turtle, Loggerhead sea turtle, Oliver ridley sea turtle). The Navy did not simulate potential exposure of sea turtles to SURTASS LFA transmissions. Because of their ecology, only the juvenile and adult stages of sea turtles could be potentially exposed to SURTASS LFA transmissions. We assume that the monitoring protocols associated with SURTASS LFA would be more effective with larger sea turtles, like adult leatherback and loggerhead turtles, than with species like olive ridley, Kemp's ridley, smaller leatherback, hawksbill, green, and some loggerhead turtles; the monitoring protocols may not detect some individual members of these species at all, which would increase their risk of exposure to sound pressure levels associated with SURTASS LFA within the mitigation zone (that is, higher than 180 dB) if they encountered SURTASS LFA vessels during a ping.

Although the probability of an interaction between SURTASS LFA sonar and individuals of any of these sea turtles is statistically small (the Navy's analyses concluded that the possible number of times a leatherback sea turtle could be in the vicinity of a SURTASS LFA sonar vessel would be less than three out of 18,000 animals per year per vessel; with the monitoring protocols, the Navy concluded that this number would approach zero), the probability could increase depending on the deployment of the SURTASS LFA vessels.

Nevertheless, sea turtles have a small probability of being exposed to SURTASS LFA transmissions. Although these species can hear low frequency sounds they have an insensitive ear. Specifically, the minimum sound turtles can hear (hearing threshold) is about 132 to 140 dB (Gentry, pers. comm., Ridgway *et al*. 1960, Barthol *et al*. 1999). Information on their behavioral response to these decibel levels is limited. However, green sea turtles were observed to avoid passing through a sound barrier created by an array of air guns with a broadband spectrum of 20-1,000 Hz; received levels were 141 to 150 dB (O'Hara and Wilcox 1990). The probability that a sea turtle would be within an ensonified area that would elicit a similar or other behavioral response is low because most of the turtles make shallow dives (984 ft [300 m] dive observed for the olive ridley sea turtle). As for the leatherback sea turtles, which can dive to depths of 3,280 ft (1000 m), the opportunity for a behavioral response is also considered to be low because 95% of their dives are less than 656 ft (200 m) deep, which would minimize their exposure to the SURTASS LFA sonar 180-dB sound field. Based on these limited sensitivities, we would not expect received levels of SURTASS LFA transmissions to reduce the reproduction, numbers, or distribution of sea turtles; as a result, these transmissions would not be expected to appreciably reduce these turtles likelihood of surviving and recovering in the wild.

Salmon. The Navy did not simulate potential exposure of Atlantic or Pacific salmon to SURTASS LFA transmissions. Because of their ecology, only the marine life stages of these salmon could be potentially exposed to SURTASS LFA transmissions depending on deployments. In the Atlantic Ocean, listed salmon have a risk of being exposed to deployments in the Gulf of Maine, in marine waters off Newfoundland, and in the southern portion of the Labrador Sea (because of operational restrictions, they would not be exposed throughout the entire Labrador Sea). In the Pacific Ocean listed salmon have a risk being exposed to deployments north of about 40° North latitude. Although salmon generally occur near the surface (within 8 to 10 meters of the surface), sockeye salmon have been caught at depths up to 61 meters while chinook salmon have been caught at depths up to 110 meters. Since the HF/M3 sonar is patterned after technology commercial and sports-fishing industries use to locate fish, we will assume that the sonar would locate schooling species like Pacific salmon, which would minimize their likelihood of being exposed to sound pressure levels in excess of 180 dB.

Nevertheless, Pacific salmon have a small probability of being exposed to SURTASS LFA transmissions. Based on their limited sensitivity to low frequency sound (Croll *et al.* 1999), we would not expect received levels of SURTASS LFA transmissions to reduce the reproduction, numbers, or distribution of these salmon; as a result, these transmissions would not be expected to appreciably reduce these salmon's likelihood of surviving and recovering in the wild.

CONCLUSION

After reviewing the current status of the threatened eastern and endangered western populations of Steller sea lions, threatened Guadalupe fur seal, endangered Hawaiian monk seal, endangered blue whale, western Pacific population of gray whales, fin whale, humpback whale, right whale, sei whale, sperm whales, threatened and endangered sea turtles, endangered species of Atlantic salmon, and threatened and endangered species of Pacific salmon, the environmental baseline for the action area, the effects of the proposed research program, and the cumulative effects, it is NMFS' biological opinion that the Navy's proposed employment of SURTASS LFA sonar in the Atlantic, Indian, and Pacific Oceans and Mediterranean Sea may adversely affect, but is not likely to jeopardize the continued existence of threatened and endangered species.

Similarly, NMFS concludes that the NMFS proposal to amend its regulations governing the take and importation of marine mammals under the MMPA to make it possible for NMFS to issue annual letters of authorization that would allow the Navy to take marine mammals incidental to the employment of SURTASS LFA sonar is not likely to jeopardize the continued existence of these endangered cetacean species.

INCIDENTAL TAKE STATEMENT

Section 9 of the Act 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. Under the terms of section 7(b)(4) and section 7(o)(2), taking that is incidental to and not intended as part of the agency action is not considered to be prohibited taking under the Act provided that such taking is in compliance with the terms and conditions of the Incidental Take Statement.

Because of the geographic scope and scale of this programmatic biological opinion NMFS cannot estimate the amount or extent of incidental take of threatened or endangered species by the proposed employment of SURTASS LFA sonar. Consequently, NMFS will identify the amount or extent of take that would be associated with the employment of SURTASS LFA when we review the annual letters of authorization for compliance with section 7 of the Endangered Species Act of 1973, as amended.

CONSERVATION RECOMMENDATIONS

Section 7(a)(1) of the ESA directs Federal agencies to utilize their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of the threatened and endangered species. Conservation recommendations are discretionary measures suggested to minimize or avoid adverse effects of a proposed action on listed species, to minimize or avoid adverse modification of critical habitat, or to develop additional information.

1) If a marine mammal is detected within the 180-dB safety zone, SURTASS LFA sonar transmissions will be immediately suspended. Transmissions should not resume earlier than 15 minutes after:
   a) all marine mammals have left the area of the 180-dB re 1 μPa_{rms} sound field; and
   b) there is no further detection of the animal within the 180 dB re 1 μPa_{rms} sound field as determined by the visual and/or passive or active acoustic monitoring.

2) The HFM3 source should be ramped-up slowly to operating levels over a period of no less than 5 minutes:
   a) No later than 30 minutes before the first SURTASS LFA sonar transmission;
   b) Prior to any SURTASS LFA sonar calibrations or testings that are not part of regular SURTASS LFA sonar transmissions; and
   c) Anytime after the HFM3 source has been powered down for a period of time greater than two minutes.

3) SURTASS LFA sonar would be operated such that the sound field does not exceed 180 dB (re 1 μPa(rms)):
   a) At a distance of 12 nautical miles (nm) (22 kilometers (km)) from any coastline, including offshore islands;
   b) Within any offshore area that has been designated as biologically important for marine mammals (see below), during the biologically important season for that particular area;
   c) Within the offshore boundaries that extend beyond 12 nm (22 km) of the following National Marine Sanctuaries: (1) Monterey Bay, (2) Gulf of the Farallones, and (3) Cordell Bank;
   d) Within 23 nm (37.4 km) during the months of December, January, March, and May of each year in the Olympic Coast National Marine Sanctuary.

4) The following areas have been designated by NMFS as offshore areas of critical biological importance for marine mammals (by season if appropriate):

Area	Location	Important Months
200-meter isobath off the North American Coast	From 28° N to 50° N, west of 40° W	Year round
Antarctic Convergence Zone	30° E to 80° N : 45° S 80° E to 150° E: 55° S 150° E to 50° W: 60° S 50° W to 30° E: 50° S	October through March
Costa Rica Dome	Centered at 9° N and 88° W	Year round
Penguin Bank	South of Molokai Island, Hawaii	November 1 through May 1

In order for NMFS to be kept informed of actions minimizing or avoiding adverse effects or benefiting listed species or critical habitat that has been designated for them, NMFS asks the U.S. Navy to notify us of the disposition of these conservation recommendations.

REINITIATION STATEMENT

This concludes formal consultation on the U.S. Navy=s proposed use of Surveillance Towed Array Sensor System Low Frequency Active Sonar, pursuant to the provisions of section 10 of the Endangered Species Act and Marine Mammal Protection Act. As provided in 50 CFR '402.16, reinitiation of formal consultation is required where discretionary Federal agency involvement or control over the action has been retained (or is authorized by law) and if: (1) the amount or extent of incidental take is exceeded; (2) new information reveals effects of the agency action that may affect 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 incidental take is exceeded, section 7 consultation must be reinitiated immediately.

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