National Marine Fisheries Service
Endangered Species Act Section 7 Consultation

Biological and Conference Opinion

Agency: NOAA's National Marine Fisheries Service-Office of Protected Resources-Permits, Conservation, and Education Division

Activities Considered: Issuance of permit to John Bengtson, National Marine Fisheries Service, National Marine Mammal Laboratory [Permit No. 14245]

Consultation Conducted by: NOAA's National Marine Fisheries Service-Office of Protected Resources-Endangered Species Division

Approved by: Helle Solvik for J H Lecky

Date: April 22, 2010 AHL 6/10/11

Section 7(a)(2) of the Endangered Species Act (ESA)(16 U.S.C. 1531 *et seq.*) requires that each federal agency shall ensure that any action authorized, funded, or carried out by such agency is not likely to jeopardize the continued existence of any endangered or threatened species or result in the destruction or adverse modification of critical habitat of such species. When the action of a federal agency "may affect" a listed species or critical habitat designated for them, that agency is required to consult with either the NOAA's National Marine Fisheries Service (NMFS) or the U.S. Fish and Wildlife Service, depending upon the listed resources that may be affected. For the actions described in this document, the action agency is the NMFS' Office of Protected Resources-Permits, Conservation, and Education Division (Permits Division), which proposes to authorize close approach, aerial survey, biopsy, and tagging activities on blue, fin, sei, bowhead, North Pacific right, humpback, sperm, Cook Inlet beluga, southern resident killer, and Hawaiian insular false killer whales in the Beaufort, Bering, and Chukchi Seas and North Pacific Ocean, as well as humpback whales in the western North Atlantic Ocean. Unintentional harassment of both eastern and western DPSs of Steller sea lions, Beringia bearded seals and, Arctic ringed seals would also be authorized. The consulting agency for these proposals is the NMFS' Office of Protected Resources – Endangered Species Division.

This document represents the NMFS' biological and conference opinion (Opinion) of the effects of the proposed actions on endangered and threatened species and designated critical habitat and has been prepared in accordance with section 7 of the ESA. This Opinion is based on information provided in the application, draft permit, environmental assessment, recovery plans for listed whales, the most current marine mammal stock assessment reports, past and current research and population dynamics modeling efforts, monitoring reports from prior research, other information provided by the applicant, and other biological opinions involving similar marine mammal research, and other information.

Consultation history

On April 27, 2010, the Permits Division published a notice in the Federal Register soliciting public comment on their intent to issue the proposed permit.

On November 22, 2010, NMFS' Endangered Species Division received a request for formal consultation from the Permits Division to authorize Permit Number 14245, John Bengtson, National Marine Fisheries Service, National Marine Mammal Laboratory.

On January 13, 2011, NMFS' Endangered Species Division initiated formal consultation with the Permits Division to authorize Permit Number 14245, John Bengtson, National Marine Fisheries Service, National Marine Mammal Laboratory.

Description of the proposed action

The NMML proposes to conduct numerous research projects on marine mammals to aide in ongoing studies of marine mammal abundance, distribution, and population dynamics. These studies include aerial and vessel-based transect surveys as well as small boat approaches of marine mammals for photoidentification, biopsy sampling, and telemetry tagging (both invasive implantable tagging, non-invasive suction-cup tagging, as well as dermal [dart/dash] tagging).

Aerial surveys are typically conducted from an Aero Commander 680 or NOAA twin otter aircraft along preplanned tracklines, but deviations to investigate and circle a marine mammal sighting typically occur. Circling can occur up to 16 times, but four to eight circling episodes are more common for target, high-priority, or unusual species sightings. Over the course of a survey season, an individual or group may be surveyed multiple times (exemplified by Cook Inlet beluga whale surveys, whose population occurs in a limited range). Flight duration is typically six hours, but can be significantly longer. Flight altitude varies based upon project, but can range from ~100-500 m and are typically flown at ~229-305 m. Speed is usually 167-222 km/hr. During this time, photographs and sighting documentation are recorded from the plane Aerial surveys would also be conducted to assist surface vessels in locating target individuals, particularly during North Pacific right whale surveys. In this situation, a survey aircraft would fly transect tracklines until a target individual is identified. The survey aircraft would circle or otherwise maintain contact with the individual until the vessel arrives, which could engage in photoidentification, biopsy sampling, and/or deployment of telemetry devices. Survey vessels may also engage in oceanographic and prey sampling in instances where target whales are observed feeding.

Vessel surveys are also conducted along pre-planned tracklines, but the vessel may deviate from its course to investigate a marine mammal sighting. Marine mammals would be followed to obtain photoidentification records, biopsy samples, or attach a telemetry device to a target individual(s). A survey vessel may be used during close approaches, or a smaller vessel may be launched from the survey vessel. Large vessels would approach an individual laterally or from behind using a minimum speed required to close the distance with the target individual or group (usually less than 10 knots) down to ~500m. Small vessels would approach from behind, with a minimum of course and speed changes, and at a speed that would not rapidly overtake a target individual or group. Approaches for photoidentification or biopsy would close to within 15-20 m for large whales or 5-10 m for smaller species. Approaches for tagging would close to within the

range that the deployment method would allow (see last paragraph of this section). Focal follows before or after biopsy or tagging attempts would not be conducted except on occasion.

Biopsy sampling would be conducted during some close approaches. All age classes except neonates could be sampled. Depending upon the species, vessel used, and other circumstances, different methods may be used to obtain a biopsy. Bowriding odontocetes may be sampled with an airgun or pole-spear, while whales may be sampled using a crossbow, airgun, or Larsen gun (modified 0.38 caliber gun firing blanks to propel a dart out of the rifle barrel). Samples are preferably obtained from just below and behind the dorsal fin, specifically avoiding the individual's facial area. Biopsy darts are roughly 25 mm in depth for small cetaceans and 40-60 mm for medium and large cetaceans. It is noteworthy that although invasive tagging of Cook Inlet beluga whales (which requires capture, handling, and restraint) was proposed by the applicant, the permit under consideration does not include these activities and are not within the scope of this consultation. Biopsy and suction cup tagging would be authorized, however, requiring close approach.

Tagging may occur using a variety of methods (pole, crossbow, Larsen gun, modified line thrower) and instrument types (implantable, dart/dash, suction cup). Suction cup tags would be applied to all age classes except neonates, while skin-penetrating tags would be applied to juveniles, subadults, and adults only. Depth of penetration into blubber would vary depending upon the tag type and target species. Medium-sized cetaceans, such as killer whales and false killer whales would not receive implantable tags (large whale only), but would instead receive dart/dash tags (Figures 1 and 2), which implant using dermal anchors (blubber implantation only) instead of blubber- and muscle-penetrating implantable tags (Figure 3). All species may receive suction-cup tags (Figure 4). Suction cups may range in size from three to 30 cm in size and use an inert silicon grease to improve attachment duration. Dart or dash tags would be deployed using a crossbow or pneumatic rifle aimed at the dorsal fin or just below it. A tether line would be fired with the dart-encasing bolt to retrieve the bolt following tag attachment. Implantable tags may be partially or fully implantable. Partially implantable tags are typically streamlined rectangles with deeply buried anchors into an individual's epaxial muscle or blubber. Fully implantable tags are typically cylinders with a penetrating conical tip on one end and a flat end where an antenna and saltwater switch is located; this later portion is usually a few centimeters above the whale's skin. Extensive efforts are made to concurrently ensure maximum longevity for designed transmission, size reduction, and functional reliability. Although these are the typical designs of cetacean tagging devices currently employed, the field of telemetry design is rapidly evolving, particularly in miniaturization, tag longevity, and inclusion of additional instrumentation. Therefore, it is reasonable to expect that new tag designs may be developed during the life of this permit and utilized by the applicants. However, we do not expect that the impacts of any new tag design to be substantially different or more significant than the impacts assessed in this consultation.

Figure 1. A dash tag prior to deployment.

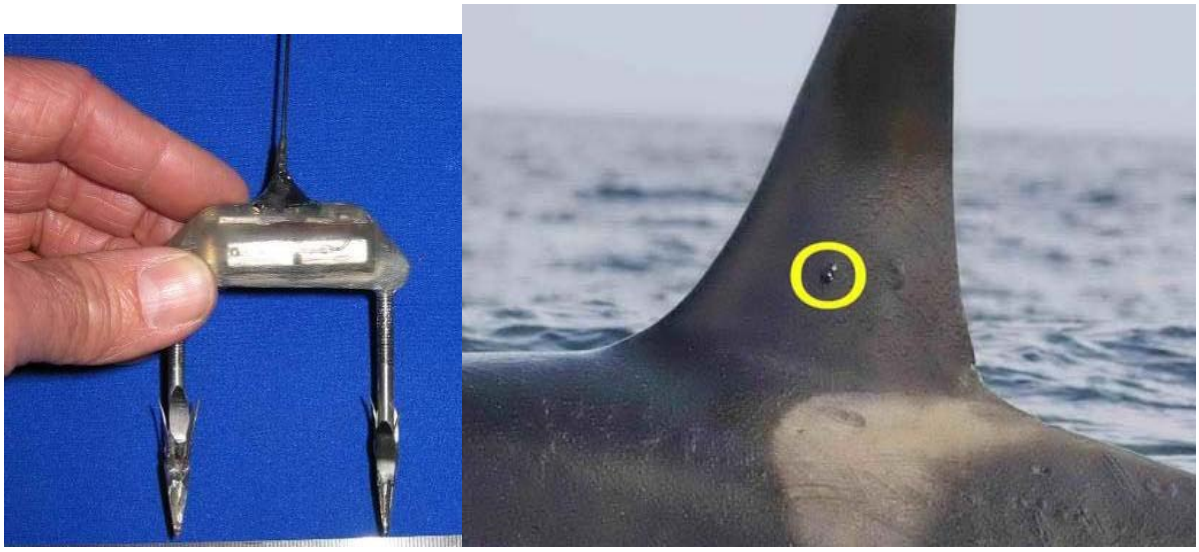

Figure 2. A dart tag prior to deployment and implanted in a killer whale's dorsal fin.

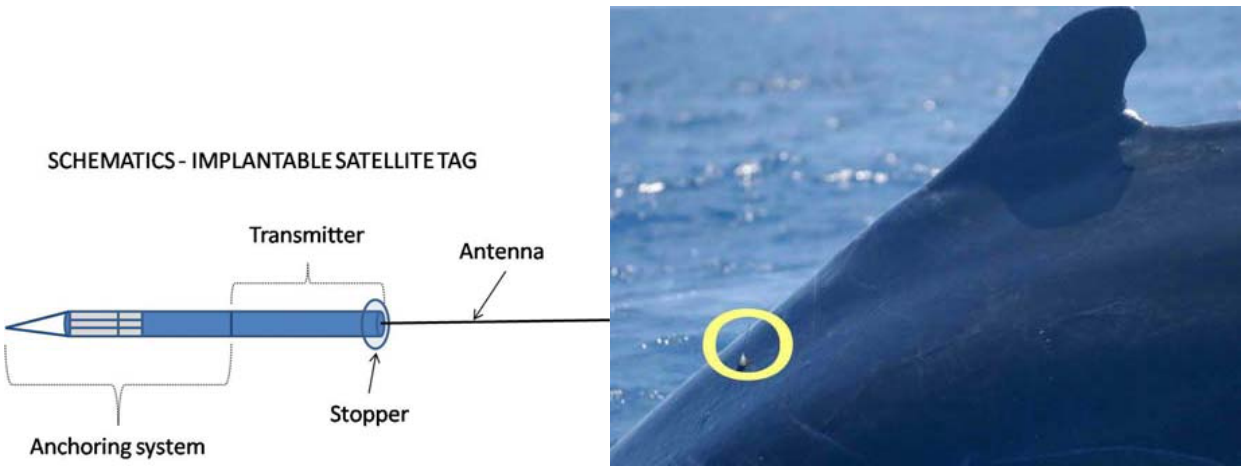

Figure 3. Schematic of an implantable tag and the blunt, antenna-protruding end of an implanted tag.

Figure 4. A suction cup tag attached to a humpback whale.

Tags generally fall within one of five categories: 1. radio transmitter: suction- cup or dart/dash device (up to 9.5 cm by 2.5 cm by 1.3 cm [42 g]) that emits signals beyond the range of a target individual's hearing (or that of its predators or prey) for hours to months; 2. time-depth recorders: recoverable instruments (up to 10 cm by 3 cm by 3 cm [42 g]) attached by suction cup for hours to a few days; 3. satellite transmitter: long-term (days to > one year), implantable device (14 cm by 9 cm by 3 cm [450 g] for implantable tags, up to 6.3 cm by 3 cm by 1.9 cm [40 g] for dart tags) that can couple with additional sensing instruments and transmit information to orbiting satellites; 4. acoustic time-depth recorder (DTAG): suction cup or dart/dash attached recoverable device (up to 9.5 cm by 2.5 cm by 1.3 cm [42 g]) that adds an acoustic recorder to the previously described time-depth recorder for attachment between hours and days; and 5. "Critter-Cam": a suction cup or dart/dash anchored video camera (up to 30 cm by 8 cm by 8 cm [<1000 g]) that records video for a period of hours.

Dash tag attachments consist of a solid-core needle <10 cm by 0.64 cm (similar in size to a biopsy plug) with raised rings around the cylinder to hamper tag removal (Figure 1). The end that penetrates the target individual is sharpened to facilitate cleaner entry, reducing the potential for infection. The actual tag is attached via a corrodible link (decays over hours to weeks based upon design). Some tags can release on command by receiving an acoustic signal. Dash tags are fired from a compressed air device with a housing that separates upon impact with the target individual. Effective range for deployment is 10-25 m from the target and the target area is from one-meter posterior of the blowhole to one to two meters cranial of the peduncle and above the water line. Once the tag detaches, the needle remains imbedded in the target individual. It is anticipated that the body will reject the needle over time, treating it as a foreign body, and pushing it out of the opening from whence it entered over the course of several days to weeks.

Dart tags are essentially miniaturized satellite transmitters (Figure 2). Tags are fired with crossbows or pneumatic rifles, with the tag being deployed with a bolt that is subsequently retrieved by an attached tether line. Attachment is via a pair of titanium or stainless steel barbed darts that penetrate up to 6.5 cm. Tags are expunged from the body over the course of weeks to a few months, although the applicant is pursuing the design of a tag that would remain attached in

excess of one year. Target region is the dorsal fin or just beneath it.

Partially implantable tags not only provide satellite telemetry data, but normally include accessory sensing instrumentation. Although the attachment system is similar to that of implantable tags (i.e., deep-penetrating barbs below the muscle-fascia-blubber interface), the instrumentation package itself remains outside the body. Attachment can last over one year. As with fully implantable tags, the target region is high on the individual's back.

Implantable tags penetrate much further into the body than do dash or dart tags, as they are designed to anchor underneath the muscle-fascia-blubber interface, giving them a much longer attachment time. Penetration depth is customized around the species, size, and body condition of the target individual. As with other penetrating tags, stoppers on the tag limit the depth to which tags can penetrate. The tags are generally designed of materials considered inert to tissue interaction, such as surgical grade stainless steel, titanium, or ultra high molecular weight surgical material. Recurved barbs or toggles along the length of the tag hamper tag expulsion, which occurs slowly over months and, in some cases, over more than two years.

Tags may be deployed either by passively awaiting a target individual to pass near the research vessel (or other natural/manmade platform on occasion) while surfacing or by active vessel approach to a target individual (more common). An individual could be targeted to carry up to two devices which pierce the skin and one additional non-piercing tag (i.e., suction cup tag). Tags can be applied using an eight m long pole at three to six meters away from the target individual, which can simultaneously apply a tag and extract a biopsy sample, and usually requires 30-45 minutes to approach and deploy. A modified line thrower (ARTS) device uses compressed air to deploy satellite tags on target individuals 10-30 m away. This system usually involves 5-30 minutes to approach and deploy a tag. Crossbows can fire tags onto target individuals 5-10 m away and usually involve the use of a tether to retrieve a bolt. The same requirement applies to a Larsen gun, which fires blanks but can be used 25-30 m from the target individual (minimum distance is 10 m).

Approach to the Assessment

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

The final steps of our analyses – establishing the risks those responses pose to listed resources –

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

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

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

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

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

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

To conduct these analyses, we rely on all of the evidence available to us. This evidence consists of monitoring reports submitted by past and present permit holders, reports from NMFS Science Centers; reports prepared by natural resource agencies in States and other countries, reports from non-governmental organizations involved in marine conservation issues, the information provided by the Permits Division when it initiates formal consultation, and the general scientific literature.

We supplement this evidence with reports and other documents – environmental assessments, environmental impact statements, and monitoring reports – prepared by other federal and state agencies like the Minerals Management Service, U.S. Coast Guard, and U.S. Navy whose operations extend into the marine environment.

During the consultation, we conducted electronic searches of the general scientific literature using search engines, including Agricola, Ingenta Connect, Aquatic Sciences and Fisheries Abstracts, JSTOR, Conference Papers Index, First Search (Article First, ECO, and WorldCat), Web of Science, Oceanic Abstracts, Google Scholar, and Science Direct.

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

The analyses used in this Opinion include several assumptions. As far as we are able to determine, field researchers cannot generally identify specific individuals in the field and, therefore, have no mechanism to know what previous exposure an individual has had to proposed activities or other natural or anthropogenic stressors. Based upon descriptions in annual NMML monitoring reports and documentation provided by the Permits Division, we assume that proposed activities will be similar to those that the NMML has conducted in the past and the level of "effort" (magnitude of time and asset resources dedicated to the proposed action) will be roughly similar to that which has previously occurred. We assume that free-ranging cetaceans (apart from Cook Inlet beluga whales) range over wide areas and although they likely occupy restricted regions for relatively brief periods (hours to days), individuals are not expected to move widely and, as far as we can predict, broadly within an oceanographic region. Although we expect that variability in reporting exists within NMML's annual reports, based upon project type and individual reporting, these reports accurately document the number of "takes" that occurred

under the MMPA and that additional, accessory data not rising to the level of "take" (observations of unusual or rare species) are also reported.

Action Area

The proposed action area includes a broad region of the North Pacific from southern California to Washington State and Alaska. Waters surrounding Hawaii would also be included in the action area. The action area extends from shore to international waters of the North Pacific Ocean, Beaufort, Bering and Chukchi seas, Arctic Ocean, and Gulf of Alaska. Actions also have the potential to occur in the Gulf of California (Mexican territorial and EEZ waters) and territorial waters of Russia, Japan, and the Philippines. Also included in the action area are waters along the U.S. eastern seaboard, from the U.S.-Canadian border to Florida as well as U.S. territories in the Caribbean Sea (Puerto Rico and the U.S. Virgin Islands). Actions are not proposed to be conducted in the EEZ or territorial waters of Canada, Mexico, or any Caribbean nation. The applicant would be permitted to conduct research during any time of year, but anticipates most research would be conducted during late spring and summer.

Status of Listed Resources

The NMFS has determined that the actions considered in this Opinion may affect species listed in Table 1, which are provided protection under the Endangered Species Act of 1973, as amended (16 U.S.C. 1531 *et seq*.).

Table 1. Listed resources in the action area. Asterisks denote critical habitat in the action area. Double Asterisks denote proposed critical habitat in the action area.

Common Name (Distinct Population Segment, Evolutionarily Significant Unit, or Subspecies)	***Scientific Name***	***Status***
Cetaceans		
Blue whale	*Balaenoptera musculus*	Endangered
Fin whale	*Balaenoptera physalus*	Endangered
Humpback whale	*Megaptera novaeangliae*	Endangered
Killer whale (Southern Resident*)	*Orcinus orca*	Endangered
Bowhead whale	*Balaena mysticetus*	Endangered
North Atlantic right whale*	*Eubalaena glacialis*	Endangered
North Pacific right whale*	*Eubalaena japonica*	Endangered
Sei whale	*Balaenoptera borealis*	Endangered
Sperm whale	*Physeter macrocephalus*	Endangered
Beluga whale (Cook Inlet)**	*Delphinapterus leucas*	Endangered
Pinnipeds		
Hawaiian monk seal*	*Monachus schauinslandi*	Endangered
Steller sea lion (Eastern*)	*Eumetopias jubatus*	Threatened
Steller sea lion (Western*)		Endangered
Marine Turtles		
Green sea turtle (Florida & Mexico's Pacific coast colonies)	*Chelonia mydas*	Endangered
Green sea turtle (All other areas*)		Threatened
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
Olive ridley sea turtle (Mexico's Pacific coast breeding colonies)	*Lepidochelys olivacea*	Endangered
Olive ridley sea turtle (All other areas)		Threatened

Common Name (Distinct Population Segment, Evolutionarily Significant Unit, or Subspecies)	***Scientific Name***	***Status***
Anadromous Fishes		
Atlantic salmon (Gulf of Maine*)	*Salmo salar*	Endangered
Chinook salmon (California Coastal*)	*Oncorhynchus tschawytscha*	Threatened
Chinook salmon (Central Valley Spring-run*)		Threatened
Chinook salmon (Lower Columbia River*)		Threatened
Chinook salmon (Upper Columbia River Spring-run*)		Endangered
Chinook salmon (Puget Sound*)		Threatened
Chinook salmon (Sacramento River Winter-run*)		Endangered
Chinook salmon (Snake River Fall-run*)		Threatened
Chinook salmon (Snake River Spring/Summer-run*)		Threatened
Chinook salmon (Upper Willamette River*)		Threatened
Chum salmon (Columbia River*)	*Oncorhynchus keta*	Threatened
Chum salmon (Hood Canal Summer-run*)		Threatened
Coho salmon (Central California Coast*)	*Oncorhynchus kisutch*	Endangered
Coho salmon (Lower Columbia River)		Threatened
Coho salmon (Southern Oregon & Northern California Coast*)		Threatened
Coho salmon (Oregon Coast*)		
Green sturgeon (Southern*)	*Acipenser medirostris*	Threatened
Shortnose sturgeon	*Acipenser brevirostrum*	Endangered
Bocaccio (Georgia Basin)	*Sebastes paucispinis*	Endangered
Yelloweye rockfish (Georgia Basin)	*Sebastes pinniger*	Threatened
Canary rockfish (Georgia Basin)	*Sebastes ruberrimus*	Threatened
Pacific eulachon**	*Thaleichthys pacificus*	Threatened
Sockeye salmon (Ozette Lake*)	*Oncorhynchus nerka*	Threatened
Sockeye salmon (Snake River*)		Endangered
Steelhead (Central California Coast*)	*Oncorhynchus mykiss*	Threatened
Steelhead (California Central Valley*)		Threatened
Steelhead (Lower Columbia River*)		Threatened
Steelhead (Middle Columbia River*)		Threatened
Steelhead (Northern California*)		Threatened
Steelhead (Puget Sound)		Threatened
Steelhead (Snake River*)		Threatened
Steelhead (South-Central California Coast*)		Threatened
Steelhead (Southern California*)		Threatened
Steelhead (Upper Columbia River*)		Threatened
Steelhead (Upper Willamette River*)		Threatened
Marine Invertebrates		
White abalone	*Haliotis sorenseni*	Endangered
Black abalone**	*Haliotis cracherodii*	Endangered
Proposed for listing		
False killer whale (Hawaii Insular)**	*Pseudorca crassidens*	Proposed Endangered

Common Name (Distinct Population Segment, Evolutionarily Significant Unit, or Subspecies)	*Scientific Name*	*Status*
Bearded seal (Beringia)	*Erignathus barbatus nauticus*	Proposed Threatened
Ringed seal (Arctic)	*Phoca hispida hispida*	Proposed Threatened
Atlantic sturgeon (Gulf of Maine)	*Asipenser oxyrinchus*	Proposed Threatened
Atlantic sturgeon (New York Bight)		Proposed Endangered

Species not considered further

Hawaiian monk seals may co-occur with proposed actions in Hawaiian waters and be exposed to the same stressors as described for listed cetaceans. Instances of shipstrike are known for Hawaiian monk seals, but extensive vessel-based effort by the NMML in the past has not resulted in a shipstrike to any pinniped. Therefore, we find the possibility of Hawaiian monk seal shipstrike to be discountable. We are unaware of a significant use of underwater sound by Hawaiian monk seals, but it is possible that this sense can be important for listed individuals in water. However, exposure to sound frequencies within the hearing range of listed individuals would be brief at most and likely insignificant to their survival, growth, or reproduction. For these reasons, we will not consider Hawaiian monk seals further in this Opinion.

Sea turtles would be exposed to these same potential stressors. As with marine mammals, sea turtles have not been documented to be struck by researchers in the area and the possibility of this occurring is discountable. Listed salmonids, rockfish, eulachon, and sturgeon may also be exposed to potential stressors from the proposed actions. Salmonids and eulachon may occur near the ocean surface, but we expect individuals to be easily capable of moving out of the direct path of even a fast-moving vessel. Sturgeon and rockfish tend to be epibenthic in marine waters and we do not expect co-occurrence with vessels at the surface. We therefore find the potential for direct strike to listed salmonids, rockfish, and sturgeon to be discountable. For these reasons, we will not consider sea turtles, salmonids, rockfish, eulachon, or sturgeon further in this Opinion.

Although listed invertebrates (black and white abalone as well as elkhorn and staghorn coral) would co-occur with the proposed actions, we cannot identify any stressors that reasonably could impact their biology. We therefore find stressors to be insignificant for these listed species and we will not consider them further in this Opinion.

We do not expect that the proposed actions will impact the primary constituent elements of critical habitat or proposed critical habitat in the action area for any listed species. We therefore find risk to critical habitat to be discountable and we will not consider them further in this Opinion.

The biology and ecology of species with anticipated exposure below informs the effects analysis for this Opinion. Summaries of the global status and trends of each species presented provide a foundation for the analysis of species as a whole.

Cetaceans

Blue whale

Description of the species. Blue whales occur primarily in the open ocean from tropical to polar waters worldwide. Blue whales are highly mobile, and their migratory patterns are not well known (Perry et al. 1999; Reeves et al. 2004). Blue whales migrate toward the warmer waters of the subtropics in fall to reduce energy costs, avoid ice entrapment, and reproduce (NMFS 1998a). Blue whales typically occur alone or in groups of up to five animals, although larger foraging aggregations of up to 50 have been reported including aggregations mixed with other rorquals such as fin whales (Corkeron et al. 1999; Shirihai 2002).

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

Population structure. Little is known about population and stock structure[1] of blue whales. Studies suggest a wide range of alternative population and stock scenarios based on movement, feeding, and acoustic data. Some suggest that as many as 10 global populations, while others suggest that the species is composed of a single panmictic population (Gambell 1979; Gilpatrick and Perryman. 2009; Reeves et al. 1998a). For management purposes, the International Whaling Commission (IWC) considers all Pacific blue whales as a single stock, whereas under the MMPA, the NMFS recognizes four stocks of blue whales: western North Pacific Ocean, eastern North Pacific Ocean, Northern Indian Ocean, and Southern Hemisphere.

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

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

distinct populations as outlined by Conway (2005).

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

North Pacific. Blue whales occur widely throughout the North Pacific. Acoustic monitoring has recorded blue whales off Oahu and the Midway Islands, although sightings or strandings in Hawaiian waters have not been reported (Barlow et al. 1997; Northrop et al. 1971; Thompson and Friedl 1982). Nishiwaki (1966) notes blue whale occurrence among the Aleutian Islands and in the Gulf of Alaska, but until recently, no one has sighted a blue whale in Alaska for some time, despite several surveys (Carretta et al. 2005b; Forney and Brownell Jr. 1996; Leatherwood et al. 1982; Stewart et al. 1987), possibly supporting a return to historical migration patterns (Anonmyous. 2009).

Blue whales are thought to summer in high latitudes and move into the subtropics and tropics during the winter (Yochem and Leatherwood 1985). Minimal data suggest whales in the western region of the North Pacific may summer southwest of Kamchatka, south of the Aleutians, and in the Gulf of Alaska, and winter in the lower latitudes of the western Pacific (Sea of Japan, the East China, Yellow, and Philippine seas) and less frequently in the central Pacific, including Hawaii (Carretta et al. 2005b; Stafford 2003; Stafford et al. 2001; Watkins et al. 2000), although this population is severely depleted or has been extirpated (Gilpatrick and Perryman. 2009). However, acoustic recordings made off Oahu showed bimodal peaks of blue whales, suggesting migration into the area during summer and winter (McDonald and Fox 1999; Thompson and Friedl 1982). In the eastern North Pacific, blue whales appear to summer off California and occasionally as far north as British Columbia, migrating south to productive areas off Mexico and as far south as the Costa Rica Dome (10° N) from June through November (Calambokidis et al. 1998; Calambokidis et al. 1990; Chandler and Calambokidis 2004; Mate et al. 1999; Reilly and Thayer 1990; Stafford et al. 1999; Wade and Friedrichsen 1979; Wade and Gerrodette 1993). However, some data indicate that some individuals may remain here year-round (Reilly and Thayer 1990; Wade and Friedrichsen 1979). The Costa Rican Dome's productivity may allow blue whales to feed during their winter calving/breeding season and not fast (Gilpatrick and Perryman. 2009; Mate et al. 1999). A blue whale tagged off Vancouver Island in 1963 was recovered a year later in just south of Kodiak Island, supporting the idea that blue whales harvested off British Columbia were en route to and from feeding areas in the Gulf of Alaska (COSEWIC 2002). One blue whale was photo-identified off the Queen Charlotte Islands in British Columbia and resighted off the Santa Barbara Channel in California, representing the first match between California and waters further north (COSEWIC 2002).

Blue whales off southern California appear to feed on dense euphausiid schools between 100-200 m below the surface (Croll et al. 1998; Fiedler et al. 1998). These concentrations of krill are associated with upwelling regions near steep topography off the continental shelf break (Croll et al. 1999). Blue whale migrations to and from California probably reflect seasonal patterns and productivity (Croll et al. 2005). Blue whales also feed in cool, offshore, upwelling-modified waters in the eastern tropical and equatorial Pacific (Palacios 1999; Reilly and Thayer 1990). Feeding areas may be associated with a greater incidence of blue whale vocalizations (Moore et al. 2002). During summer, blue whales calls in water of the Northwest Pacific were closely associated with cold water and sharp sea surface temperature gradients or fronts, probably corresponding to zooplankton concentrations. From fall through spring, call locations were concentrated primarily near seamounts (Moore et al. 2002).

Indian Ocean. Blue whale sightings have occurred in the Gulf of Aden, Persian Gulf, Arabian Sea, and across the Bay of Bengal to Burma and the Strait of Malacca (Clapham et al. 1999; Mikhalev 1997; Mizroch et al. 1984).

Southern Hemisphere. Blue whales range from the edge of the Antarctic pack ice (40°-78°S) during the austral summer north to Ecuador, Brazil, South Africa, Australia, and New Zealand during the austral winter (Shirihai 2002). Occurrence in Antarctic waters appears to be highest from February-May as well as in November (Gedamke and Robinson. 2010; Sirovic et al. 2009). Gedamke and Robinson (2010) found blue whales to be particularly numerous and/or vocal north of Prydz Bay, Antarctica based upon sonar buoy deployments. Pygmy blue whales were also frequently heard in Antarctic waters, further south than they had previously been documented (Gedamke and Robinson 2010). Blue whales are occasionally sighted in pelagic waters off the western coast of Costa Rica and Nicaragua, near the Galápagos Islands, and along the coasts of Ecuador and northern Peru (Aguayo 1974; Clarke 1980b; Donovan 1984; LGL Ltd. 2007; Mate et al. 1999; Palacios 1999; Reilly and Thayer 1990). Individuals here may represent to populations; the true and pygmy blue whales of the Southern Hemisphere (Gilpatrick and Perryman. 2009). Although, recent analyses of vocalizations and photos have linked blue whales found in the Costa Rica Dome to the North Pacific population (Chandler and Calambokidis 2004).

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

Reproduction. Gestation takes 10-12 months, followed by a 6-7 month nursing period. Sexual maturity occurs at 5-15 years of age and calves are born at 2-3 year intervals (COSEWIC 2002; NMFS 1998b; Yochem and Leatherwood 1985). Recent data from illegal Russian whaling for Antarctic and pygmy blue whales support sexual maturity at 23 m and 19-20 m, respectively (Branch and Mikhalev 2008).

Movement. Satellite tagging indicates that, for blue whales tagged off Southern California, movement is more linear and faster (3.7 km/h) while traveling versus while foraging (1.7 km/h) (Bailey et al. 2009). Residency times in what are likely prey patches averages 21 days and constituted 29% of an individual's time overall, although foraging could apparently occur at any time of year for tagged individuals (Bailey et al. 2009). Broad scale movements also varied greatly, likely in response to oceanographic conditions influencing prey abundance and distribution (Bailey et al. 2009).

Feeding. Data indicate that some summer feeding takes place at low latitudes in upwelling-modified waters, and that some whales remain year-round at either low or high latitudes (Clarke and Charif 1998; Hucke-Gaete et al. 2004; Reilly and Thayer 1990; Yochem and Leatherwood 1985). One population feeds in California waters from June to November and migrates south in winter/spring (Calambokidis et al. 1990; Mate et al. 1999). Prey availability likely dictates blue whale distribution for most of the year (Burtenshaw et al. 2004; Clapham et al. 1999; Sears 2002 as cited in NMFS 2006a). The large size of blue whales requires higher energy requirements than smaller whales and potentially prohibits fasting Mate et al. (1999). Krill are the primary prey of blue whales in the North Pacific (Kawamura 1980; Yochem and Leatherwood 1985).

While feeding, blue whales show slowed and less obvious avoidance behavior then when not feeding (Sears et al. 1983 as cited in NMFS 2005c).

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

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

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

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

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

Direct studies of blue whale hearing have not been conducted, but it is assumed that blue whales can hear the same frequencies that they produce (low-frequency) and are likely most sensitive to

this frequency range (Ketten 1997; Richardson et al. 1995c).

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

Table 2 contains historic and current estimates of blue whales by region. Globally, blue whale abundance has been estimated at between 5,000-13,000 animals (COSEWIC 2002; Yochem and Leatherwood 1985); a fraction of the 200,000 or more that are estimated to have populated the oceans prior to whaling (Maser et al. 1981; U.S. Department of Commerce 1983).

North Atlantic. Commercial hunting had a severe effect on blue whales, such that they remain rare in some formerly important habitats, notably in the northern and northeastern North Atlantic (Sigurjónsson and Gunnlaugsson 1990). Sigurjónsson and Gunnlaugsson (1990) estimated that at least 11,000 blue whales were harvested from all whaling areas from the late nineteenth to mid-twentieth centuries. The actual size of the blue whale population in the North Atlantic is uncertain, but estimates range from a few hundred individuals to about 2,000 (Allen 1970; Mitchell 1974a; Sigurjónsson 1995; Sigurjónsson and Gunnlaugsson 1990). Current trends are unknown, although an increasing annual trend of 4.9% annually was reported for 1969–1988 off western and southwestern Iceland (Sigurjónsson and Gunnlaugsson 1990). Sigurjónsson and Gunnlaugsson (1990) concluded that the blue whale population had been increasing since the late 1950s. In the northeastern Atlantic, blue whales are most common west and south of Iceland and may be the largest concentration of blue whales in the North Atlantic (Pike et al. 2009b). In this area, the population may be recovering at a rate of 4-5%(Pike et al. 2009b)

North Pacific. Estimates of blue whale abundance are uncertain. Prior to whaling, Gambell (1976) reported there may have been as many as 4,900 blue whales. Blue whales were hunted in the Pacific Ocean, where approximately 5,761 killed from 1889–1965 (Perry et al. 1999). The IWC banned commercial whaling in the North Pacific in 1966, although Soviet whaling continued after the ban. In the eastern North Pacific, the minimum stock (based upon surveys in U.S. EEZ waters) is thought to be 1,384 whales, but no minimum estimate has been established (Carretta et al. 2006). Although blue whale abundance has likely increased since its protection in 1966, the possibility of unauthorized harvest by Soviet whaling vessel, incidental ship strikes, and gillnet mortalities make this uncertain.

Calambokidis and Barlow (2004) estimated roughly 3,000 blue whales inhabit waters off California, Oregon, and Washington based on line-transect surveys and 2,000 based on capture-recapture methods. Carretta et al.(2006) noted that the best estimate of abundance off California, Oregon, and Washington is an average of line-transect and capture-recapture estimates (1,744). Barlow (2003) reported mean group sizes of 1.0–1.9 during surveys off California, Oregon, and Washington. A density estimate of 0.0003 individuals/km^2 was given for waters off Oregon/Washington, and densities off California ranged from 0.001-0.0033 individuals/km^2 (Barlow 2003).

Table 2. Summary of past and present blue whale abundance.

Region	Population, stock, or study area	Pre-exploitation estimate	95% C.I.	Current estimate	95% C.I.	Source
Global	--	200,000	--	11,200-13,000	--	(DOC 1983; Maser et al. 1981)
				5,000-12,000		(COSEWIC 2002)
North Atlantic	Basinwide	1,100-1,500	--	100-555	--	(Braham 1991; Gambell 1976)
	NMFS - Western North Atlantic stock	--	--	308	--	(Sears et al. 1987)
North Pacific	Basinwide	4,900	--	1,400-1,900	--	(Gambell 1976)
				3,300	--	(Wade and Gerrodette 1993) and (Barlow 1997b) as combined in (Perry et al. 1999)
	Eastern Tropical Pacific	--	--	1,415	1,078-2,501	(Wade and Gerrodette 1993)
	EEZ of Costa Rica	--	--	48	22-102*	(Gerrodette and Palacios 1996)
	EEZs of Central America north of Costa Rica	--	--	94	34-257*	(Gerrodette and Palacios 1996)
	Eastern North Pacific	--	--	2,997	2,175-3,819*	(Calambokidis and Barlow 2004)
	NMFS - western North Pacific stock	--	--	n/a	--	(Carretta et al. 2006)
	NMFS - eastern North Pacific stock	--	--	1,368	CV=0.22	(Carretta et al. 2008)
Southern Hemisphere	Basinwide	150,000-210,000	--	5,000-6,000	--	(Gambell 1976; Yochem and Leatherwood 1985)
		300,000	--	--	--	(COSEWIC 2002)
		--	--	400-1,400	400-1,400	IWC, for years 1980-2000
		--	--	1,700	860-2,900	(IWC 2005c), point estimate for 1996
	Within IWC survey areas	--	--	1,255	--	(IWC 1996)
	Pygmy blue whale population	10,000	--	5,000	--	(Gambell 1976)
		13,000	--	6,500	--	(Zemsky and Sazhinov 1982)

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

Southern Hemisphere. Estimates of 4-5% for an average rate of population growth have been proposed (Yochem and Leatherwood 1985). However, a recent estimate of population growth for Antarctic blue whales throughout was 7.3% (Branch et al. 2007). Branch et al. (2007) also included an estimate of 1,700 individuals south of 60º. Blue whales in the region remain severely depleted with the 1996 estimate only 0.7% of pre-whaling levels (IWC 2005).

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

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

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

Ship strike is presently a concern for blue whale recovery. Ship strikes have recently averaged roughly one every other year (eight ship strike incidents are known Jensen and Silber (2004), but in September 2007, ships struck five blue whales within a few-day period off southern California (Calambokidis pers. comm. 2008)(Berman-Kowalewski et al. 2010). Dive data support a surface-oriented behavior during nighttime that would make blue whales particularly vulnerable to ship strikes. There are concerns that, like right whales, blue whales may surface when approached by large vessels; a behavior that would increase their likelihood of being struck. Protective measures are not currently in place. In the California/Mexico stock, annual incidental mortality due to ship strikes averaged one whale every 5 years, but we cannot determine if this reflects the actual number of blue whales struck and killed by ships (i.e., individuals not observed when struck and those who do not strand; Barlow et al. (1997; Berman-Kowalewski et al. 2010). It is believed that the vast majority of ship strike mortalities are never identified, and that actual mortality is higher than currently documented.

Increasing oceanic noise may impair blue whale behavior. Although available data do not presently support traumatic injury from sonar, the general trend in increasing ambient low-frequency noise in the deep oceans of the world, primarily from ship engines, could impair the ability of blue whales to communicate or navigate through these vast expanses (Aburto et al. 1997; Clark 2006).

There is a paucity of contaminant data regarding blue whales. Available information indicates that organochlorines, including dichloro-diphenyl-trichloroethane (DDT), polychlorinated biphenyls (PCB), benzene hexachloride (HCH), hexachlorobenzene (HCB), chlordane, dieldrin, methoxychlor, and mirex have been isolated from blue whale blubber and liver samples (Gauthier et al. 1997b; Metcalfe et al. 2004). Contaminants transfer between mother and calf meaning that young often start life with concentrations of contaminants equal to their mothers, before accumulating additional contaminant loads during life and passing higher loads to the next generation (Gauthier et al. 1997a; Metcalfe et al. 2004).

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

Fin whale

Description of the species. The fin whale is the second largest baleen whale and is widely distributed in the world's oceans. Most fin whales in the Northern Hemisphere migrate seasonally from Antarctic feeding areas in the summer to low-latitude breeding and calving grounds in winter. Fin whales tend to avoid tropical and pack-ice waters, with the high-latitude

limit of their range set by ice and the lower-latitude limit by warm water of approximately 15° C (Sergeant 1977). Fin whale concentrations generally form along frontal boundaries, or mixing zones between coastal and oceanic waters, which corresponds roughly to the 200 m isobath (the shelf edge; (Cotte et al. 2009; Nasu 1974).

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

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

North Atlantic. Fin whales are common off the Atlantic coast of the U.S. in waters immediately off the coast seaward to the continental shelf (about the 1,800 m contour).

Fin whales occur during the summer from Baffin Bay to near Spitsbergen and the Barents Sea, south to Cape Hatteras in North Carolina and off the coasts of Portugal and Spain (Rice 1998a). In areas north of Cape Hatteras, fin whales account for about 46% of the large whales observed in surveys between 1978-1982 (CETAP 1982). Little is known about the winter habitat of fin whales, but in the western North Atlantic, the species has been found from off Newfoundland south to the Gulf of Mexico and Greater Antilles, and in the eastern North Atlantic the winter range extends from the Faroes and Norway south to the Canary Islands. Fin whales in the eastern North Atlantic have been found in highest densities in the Irminger Sea between Iceland and Greenland (Víkingsson et al. 2009). In the Atlantic Ocean, a general migration in the fall from the Labrador and Newfoundland region, south past Bermuda, and into the West Indies has been theorized (Clark 1995). Historically, fin whales were by far the most common large whale found off Portugal (Brito et al. 2009).

Fin whales are also endemic to the Mediterranean Sea, where (at least in the western Mediterranean), individuals tend to aggregate during summer and disperse in winter over large spatial scales (Cotte et al. 2009). Mediterranean fin whales are genetically distinct from fin whales in the rest of the North Atlantic at the population level (Berube et al. 1999). Individuals also tend to associate with colder, saltier water, where steep changes in temperature occurred, and where higher northern krill densities would be expected (Cotte et al. 2009). A genetically distinct population resides year-round in the Ligurian Sea (IWC 2006a).

North Pacific. Fin whales undertake migrations from low-latitude winter grounds to high-latitude summer grounds and extensive longitudinal movements both within and between years (Mizroch et al. 1999). Fin whales are sparsely distributed during November-April, from 60°N, south to the northern edge of the tropics, where mating and calving may take place (Mizroch et al. 1999). However, fin whales have been sighted as far north as 60°N throughout winter (Mizroch et al. 1999). A resident fin whale population may exist in the Gulf of California

(Tershy et al. 1993).

Fin whales are observed year-round off central and southern California with peak numbers in the summer and fall (Barlow 1997b; Dohl et al. 1983; Forney et al. 1995). Peak numbers of fin whales are seen during the summer off Oregon, and in summer and fall in the Gulf of Alaska and southeastern Bering Sea (Moore et al. 2000c; Perry et al. 1999). Fin whales are observed feeding in Hawaiian waters during mid-May, and their sounds have been recorded there during the autumn and winter (Balcomb 1987; Northrop et al. 1968; Shallenberger 1981b; Thompson and Friedl 1982). Fin whales in the western Pacific winter in the Sea of Japan, the East China, Yellow, and Philippine seas (Gambell 1985a).

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

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

Reproduction. Fin whales reach sexual maturity between 5-15 years of age (COSEWIC 2005; Gambell 1985a; Lockyer 1972). Mating and calving occurs primarily from October-January, gestation lasts ~11 months, and nursing occurs for 6-11 months (Boyd et al. 1999; Hain et al. 1992). The average calving interval in the North Atlantic is estimated at about 2-3 years (Agler et al. 1993; Christensen et al. 1992a). The location of winter breeding grounds is uncertain but mating is assumed to occur in pelagic mid-latitude waters (Perry et al. 1999). Although seasonal migration occurs between presumed foraging and breeding locations, fin whales have been acoustically detected throughout the North Atlantic Ocean and Mediterranean Sea year-round, implying that not all individuals follow a set migratory pattern (Notarbartolo-Di-Sciara et al. 1999).

Feeding. Fin whales in the North Atlantic eat pelagic crustaceans (mainly krill and schooling fish such as capelin, herring, and sand lance (Borobia and Béland 1995; Christensen et al. 1992a; Hjort and Ruud 1929; Ingebrigtsen 1929; Jonsgård 1966; Mitchell 1974b; Overholtz and Nicolas 1979; Sergeant 1977; Shirihai 2002; Watkins et al. 1984a). In the North Pacific, fin whales also prefer euphausiids and large copepods, followed by schooling fish such as herring, walleye Pollock, and capelin (Kawamura 1982a; Kawamura 1982b; Ladrón De Guevara et al. 2008; Nemoto 1970; Paloma et al. 2008). Fin whales frequently forage along cold eastern boundaries of currents (Perry et al. 1999). Antarctic fin whales feed on krill, *Euphausia superba*, which occurs in dense near-surface schools (Nemoto 1959). However, off the coast of Chile, fin whales are known to feed on the euphausiid *E. mucronata* (Antezana 1970; Perez et al. 2006). Feeding may occur in waters as shallow as 10 m when prey are at the surface, but most foraging is observed in high-productivity, upwelling, or thermal front marine waters (Gaskin 1972; Nature Conservancy Council 1979 as cited in ONR 2001; Panigada et al. 2008; Sergeant 1977). While foraging, fin whales in the Mediterranean Sea have been found to move in restricted territories in a convoluted manner (Lafortuna et al. 1999).

Diving. The amount of time fin whales spend at the surface varies. Some authors have reported that fin whales make 5-20 shallow dives, each of 13-20 s duration, followed by a deep dive of 1.5-15 min (Gambell 1985a; Lafortuna et al. 2003; Stone et al. 1992). Other authors have reported that the fin whale's most common dives last 2-6 min (Hain et al. 1992; Watkins 1981a). The most recent data support average dives of 98 m and 6.3 min for foraging fin whales, while non-foraging dives are 59 m and 4.2 min (Croll et al. 2001a). However, Lafortuna et al. (1999) found that foraging fin whales have a higher blow rate than when traveling. Foraging dives in excess of 150 m are known (Panigada et al. 1999). In waters off the U.S. Atlantic Coast, individuals or duos represented about 75% of sightings during the Cetacean and Turtle Assessment Program (Hain et al. 1992). Individuals or groups of less than five individuals represented about 90% of the observations. Barlow (2003) reported mean group sizes of 1.1–4.0 during surveys off California, Oregon, and Washington.

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

Although their function is still in doubt, low-frequency fin whale vocalizations travel over long distances and may aid in long-distance communication (Edds-Walton 1997; Payne and Webb 1971). During the breeding season, fin whales produce pulses in a regular repeating pattern, which have been proposed to be mating displays similar to those of humpbacks (Croll et al. 2002). These vocal bouts last for a day or longer (Tyack 1999).

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

Status and trends. Fin whales were originally listed as endangered in 1970 (35 FR 18319), and this status continues since the inception of the ESA in 1973. Although fin whale population structure remains unclear, various abundance estimates are available (Table 3). Pre-exploitation fin whale abundance is estimated at 464,000 individuals worldwide; the estimate for 1991 was roughly 25% of this (Braham 1991). Historically, worldwide populations were severely depleted by commercial whaling, with more than 700,000 whales harvested in the twentieth century (Cherfas 1989).

Table 3. Summary of past and present fin whale abundance.

Region	Population, stock, or study area	Pre-exploitation estimate	95% C.I.	Current estimate	95% C.I.	Source
Global	--	>464,000	--	119,000	--	(Braham 1991)
North Atlantic						
	Basinwide	30,000-50,000	--	--	--	(Sergeant 1977)
		360,000	249,000-481,000	--	--	(Roman and Palumbi 2003)
	Central and Northeastern Atlantic	--	--	30,000	23,000-39,000	(IWC 2007)
	Western North Atlantic	--	--	3,590-6,300	--	(Braham 1991)
	NMFS - Western North Atlantic stock	--	--	2,269	CV=0.37	(NMFS 2008d)
	Northeastern U.S. Atlantic Continental Shelf	--	--	2,200-5,000	--	(Hain et al. 1992; Waring et al. 2000)
	IWC - Newfoundland-Labrador stock	--	--	13,253	0-50,139*	(IWC 1992b)
	IWC - British Isles-Spain and Portugal stock	10,500	9,600-11,400	4,485	3,369-5,600	(Braham 1991)
				17,355	10,400-28,900	(Buckland et al. 1992)
	IWC - North Norway stock	--	--	--	--	--
	IWC - East Greenland-Iceland stock	--	--	11,563	5,648-17,478*	(Gunnlaugsson and Sigurjónsson 1990)
	IWC - West Greenland stock	--	--	1,700	840-3,500	(IWC 2006a)
North Pacific	Basinwide	42,000-45,000	--	16,625	14,620-18,630	(Braham 1991; Ohsumi and Wada 1974)
	Central Bering Sea	--	--	4,951	2,833-8,653	(Moore et al. 2002)
	NMFS - Northeast Pacific stock, west of Kenai Peninsula	--	--	5,700	--	(Angliss and Allen 2007)
	NMFS - California/Oregon/ Washington stock	--	--	2,636	CV=0.15	(Carretta et al. 2008)
	NMFS - Hawaii stock	--	--	174	0-420*	(Carretta et al. 2008)
Southern Hemisphere	Basinwide	400,000	--	85,200	--	(Braham 1991; IWC 1979)
	South of 60 ^{o}S	--	--	1,735	514-2,956	(IWC 1996)
	South of 30 ^{o}S	--	--	15,178	--	(IWC 1996)
	Scotia Sea and Antarctic Peninsula	--	--	4,672	792-8,552*	(Hedley et al. 2001; Reilly et al. 2004)

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

North Atlantic. Sigurjónsson (1995) estimated that between 50,000 and 100,000 fin whales once populated the North Atlantic, although he provided no data or evidence to support that estimate. However, over 48,000 fin whales were caught between 1860- 1970 (Braham 1991). Although protected by the IWC, from 1988-1995 there have been 239 fin whales harvested from the North Atlantic. Recently, Iceland resumed whaling of fin whales despite the 1985 moratorium imposed by the IWC. Forcada et al. (1996) estimated that 3,583 individuals (95% CI = 2,130- 6,027) inhabit the western Mediterranean Sea. Goujon et al. (1994) estimated 7,000-8,000 fin whales in the Bay of Biscay. Vikingsson et al. (2009) estimated roughly 20,000 fin whales to be present in a large portion of the eastern North Atlantic in 1995, which increased to roughly 25,000 in 2001. The authors concluded that actual numbers were likely higher due to negative bias in their analysis, and that the population(s) were increasing at 4% annually

(Víkingsson et al. 2009).

North Pacific. The status and trend of fin whale populations is largely unknown. Over 26,000 fin whales were harvested between 1914-1975 (Braham 1991 as cited in Perry et al. 1999). NMFS estimates roughly 3,000 individuals occur off California, Oregon, and Washington based on ship surveys in summer/autumn of 1996, 2001, and 2005, of which estimates of 283 and 380 have been made for Oregon and Washington alone (Barlow 2003; Barlow and Taylor 2001; Forney 2007). Barlow (2003) noted densities of up to 0.0012 individuals/km^2 off Oregon and Washington and up to 0.004 individuals/km^2 off California.

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

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

Anthropogenic threats. Fin whales have undergone significant exploitation, but are currently protected under the IWC. Fin whales are still hunted in subsistence fisheries off West Greenland. In 2004, five males and six females were killed, and two other fin whales were struck and lost. In 2003, two males and four females were landed and two others were struck and lost (IWC 2005). Between 2003 and 2007, the IWC set a catch limit of up to 19 fin whales in this subsistence fishery. However, the scientific recommendation was to limit the number killed to four individuals until accurate populations could be produced (IWC 2005). In the Antarctic Ocean, fin whales are hunted by Japanese whalers who have been allowed to kill up to 10 fin whales each ear for the 2005-2006 and 2006-2007 seasons under an Antarctic Special Permit NMFS (2006d). The Japanese whalers plan to kill 50 whales per year starting in the 2007-2008 season and continuing for the next 12 years (IWC 2006b; Nishiwaki et al. 2006).

Fin whales experience significant injury and mortality from fishing gear and ship strikes (Carretta et al. 2007a; Douglas et al. 2008; Lien 1994; Perkins and Beamish 1979; Waring et al. 2007). Between 1969-1990, 14 fin whales were captured in coastal fisheries off Newfoundland and Labrador; of these seven are known to have died because of capture (Lien 1994; Perkins and Beamish 1979). In 1999, one fin whale was reported killed in the Gulf of Alaska pollock trawl fishery and one was killed the same year in the offshore drift gillnet fishery (Angliss and Outlaw 2005a; Carretta et al. 2004a). According to Waring et al. (2007), four fin whales in the western North Atlantic died or were seriously injured in fishing gear, while another five were killed or injured as a result of ship strikes between January 2000 and December 2004. Jensen and Silber (2004) review of the NMFS' ship strike database revealed fin whales as the most frequently confirmed victims of ship strikes (26% of the recorded ship strikes [n = 75/292 records]), with

most collisions occurring off the east coast, followed by the west coast of the U.S. and Alaska/Hawaii. Between 1999-2005, there were 15 reports of fin whales strikes by vessels along the U.S. and Canadian Atlantic coasts (Cole et al. 2005a; Nelson et al. 2007a). Of these, 13 were confirmed, resulting in the deaths of 11 individuals. Five of seven fin whales stranded along Washington State and Oregon showed evidence of ship strike with incidence increasing since 2002 (Douglas et al. 2008). Similarly, 2.4% of living fin whales from the Mediterranean show ship strike injury and 16% of stranded individuals were killed by vessel collision (Panigada et al. 2006). There are also numerous reports of ship strikes off the Atlantic coasts of France and England (Jensen and Silber 2004).

Management measures aimed at reducing the risk of ships hitting right whales should also reduce the risk of collisions with fin whales. In the Bay of Fundy, recommendations for slower vessel speeds to avoid right whale ship strike appear to be largely ignored (Vanderlaan et al. 2008). However, new rules for seasonal (June through December) slowing of vessel traffic to 10 knots and changing shipping lanes by less than one nautical mile to avoid the greatest concentrations of right whales are predicted to be capable of reducing ship strike mortality by 27% in the Bay of Fundy region.

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

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

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

Sei whale

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

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

population or several discrete ones.

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

North Pacific. Some mark-recapture, catch distribution, and morphological research indicate more than one population may exist – one between 155°-175° W, and another east of 155° W (Masaki 1976; Masaki 1977). 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 and south to California to the east and Japan and Korea to the west (Leatherwood et al. 1982; Nasu 1974). Sightings have also occurred in Hawaiian waters (Smultea et al. 2010). Sei whales have been occasionally reported from the Bering Sea and in low numbers on the central Bering Sea shelf (Hill and DeMaster 1998). Whaling data suggest that sei whales do not venture north of about 55°N (Gregr et al. 2000). Masaki (1977) reported sei whales concentrating in the northern and western Bering Sea from July-September, although other researchers question these observations because no other surveys have reported sei whales in the northern and western Bering Sea. Horwood (1987) evaluated Japanese sighting data and concluded that sei whales rarely occur in the Bering Sea. Horwood (1987) reported that 75-85% of the North Pacific population resides east of 180°. During winter, sei whales are found from 20°-23° N (Gambell 1985b; Masaki 1977). Considering the many British Columbia whaling catches in the early to mid 1900s, sei whales have clearly utilized this area in the past (Gregr et al. 2000; Pike and MacAskie 1969).

Southern Hemisphere. Sei whales occur throughout the Southern Ocean during the austral summer, generally between 40°-50° S (Gambell 1985b). During the austral winter, sei whales occur off Brazil and the western and eastern coasts of southern Africa and Australia, although all of the 20 sightings off Argentina occurred in August or September (Iniguez et al. 2010). However, sei whales generally do not occur north of 30º S in the Southern Hemisphere (Reeves et al. 1999). However, confirmed sighting records exist for Papua New Guinea and New Caledonia, with unconfirmed sightings in the Cook Islands (Programme) 2007). A sei whale stranded in New Caledonia during May of 1962 (Borsa 2006).

In the Southern Hemisphere, the IWC has divided the Southern Ocean into six baleen whale feeding areas – designated at 60° S latitude and longitude as: 60°-120° W (Area I), 0°-60° W (Area II), 0° to 70° E (Area III), 70°-130° E (Area IV), 130°-170° W (Area V), and 170°-120°W (Area VI).

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

Reproduction. Reproductive activities for sei whales occur primarily in winter. Gestation is about 12.7 months, calves are weaned at 6-9 months, and the calving interval is about 2-3 years (Gambell 1985b; Rice 1977). Sei whales become sexually mature at about age 10 (Rice 1977).

Feeding. Sei whales are primarily planktivorous, feeding mainly on euphausiids and copepods, although they are also known to consume fish (Waring et al. 2006). In the Northern Hemisphere, sei whales consume small schooling fish such as anchovies, sardines, and mackerel when locally abundant (Mizroch et al. 1984; Rice 1977). Sei whales in the North Pacific feed on euphausiids and copepods, which make up about 95% of their diets (Calkins 1986). The dominant food for sei whales off California during June-August is northern anchovy, while in September-October whales feed primarily on krill (Rice 1977). 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). In the Southern Ocean, analysis of stomach contents indicates sei whales consume *Calanus* spp. and small-sized euphasiids with prey composition showing latitudinal trends (Kawamura 1974). Evidence indicates that sei whales in the Southern Hemisphere reduce direct interspecific competition with blue and fin whales by consuming a wider variety of prey and by arriving later to feeding grounds (Kirkwood 1992). 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. In the North Pacific, sei whales appear to prefer feeding along the cold eastern currents (Perry et al. 1999).

Vocalization and hearing. Data on sei whale vocal behavior is limited, but includes records off the Antarctic Peninsula of broadband sounds in the 100-600 Hz range with 1.5 s duration and tonal and upsweep calls in the 200-600 Hz range of 1-3 s durations (McDonald et al. 2005). Differences may exist in vocalizations between ocean basins (Rankin and Barlow 2007). Vocalizations from the North Atlantic consisted of paired sequences (0.5-0.8 sec, separated by 0.4-1.0 sec) of 10-20 short (4 msec) FM sweeps between 1.5-3.5 kHz (Thomson and Richardson 1995).

Status and trends. The sei whale was originally listed as endangered in 1970 (35 FR 18319), and this status remained since the inception of the ESA in 1973. Table 4 provides estimates of historic and current abundance for ocean regions.

Table 4. Summary of past and present sei whale abundance.

Region	Population, stock, or study area	Pre-exploitation estimate	95 % C.I.	Current estimate	95% C.I.	Source
Global	--	>105,000	--	25,000	--	(Braham 1991)
North Atlantic	Basinwide	--	--	>4000	--	(Braham 1991)
	NMFS - Nova Scotia stock	--	--	207	--	(NMFS 2008d)
	IWC - Iceland-Denmark stock	--	--	1,290	0-2,815*	(Cattanach et al. 1993)
	IWC - Iceland-Denmark stock	--	--	1,590	343-2,837*	(Cattanach et al. 1993)
North Pacific	Basinwide	42,000	--	7,260-12,620*	--	(Tillman 1977); *circa 1974
	NMFS - eastern North Pacific stock	--	--	46	CV=0.61	(Carretta et al. 2008)
	NMFS - Hawaii stock	--	--	77	0-237*	(Carretta et al. 2008)
Southern Hemisphere	Basinwide	63,100	--	--	--	(Mizroch et al. 1984)
	Basinwide	65,000	--	--	--	(Braham 1991)
	South of 60°S	--	--	626	553-699	(IWC 1996)
	South of 30°S	--	--	9,718	--	(IWC 1996)

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

North Atlantic. No information on sei whale abundance exists prior to commercial whaling (Perry et al. 1999). Between 1966 and 1972, whalers from land stations on the east coast of Nova Scotia engaged in extensive hunts of sei whales on the Nova Scotia shelf, killing about 825 sei whales (Mitchell and Chapman 1977). In 1974, the North Atlantic stock was estimated to number about 2,078 individuals, including 965 whales in the Labrador Sea group and 870 whales in the Nova Scotia group (Mitchell and Chapman 1977). In the northwest Atlantic, Mitchell and Chapman (1977) estimated the Nova Scotia stock to contain between 1,393-2,248 whales; and an aerial survey program conducted from 1978 to 1982 on the continental shelf and edge between Cape Hatteras, North Carolina, and Nova Scotia generated an estimate of 280 sei whales (CETAP 1982). These two estimates are more than 20 years out of date and likely do not reflect the current true abundance; in addition, the Cetacean and Turtle Assessment Program estimate has a high degree of uncertainty and is considered statistically unreliable (Perry et al. 1999; Waring et al. 2004b; Waring et al. 1999). The total number of sei whales in the U.S. Atlantic EEZ remains unknown (Waring et al. 2006). Rice (1977) estimated total annual mortality for adult females as 0.088 and adult males as 0.103.

North Pacific. Ohsumi and Fukuda (1975) estimated that sei whales in the North Pacific numbered about 49,000 whales in 1963, had been reduced to 37,000-38,000 whales by 1967, and reduced again to 20,600-23,700 whales by 1973. From 1910-1975, 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 1911-1955. 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. 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 1968-1969, after which the sei whale population declined rapidly (Mizroch et al. 1984). When commercial whaling for sei whales ended in 1974, the population in the North Pacific had been reduced to 7,260-12,620 animals (Tillman 1977). There have been no direct estimates of sei whale populations for the eastern Pacific Ocean (or the entire Pacific). During aerial surveys between 1991 and 2001, there were two confirmed sightings of sei whales along the U.S. Pacific coast. The minimum estimate of individuals along the U.S. west coast between 1996-2001 was 35 (Carretta et al. 2006).

Natural threats. The foraging areas of right and sei whales in the western North Atlantic Ocean overlap and both whales feed preferentially on copepods (Mitchell 1975).

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

Anthropogenic threats. Human activities known to threaten sei whales include whaling, commercial fishing, and maritime vessel traffic. Historically, whaling represented the greatest threat to every population of sei whales and was ultimately responsible for listing sei whales as an endangered species. Sei whales are thought to not be widely hunted, although harvest for scientific whaling or illegal harvesting may occur in some areas.

Sei whales are occasionally killed in collisions with vessels. Of three sei whales that stranded along the U.S. Atlantic coast between 1975-1996, two showed evidence of collisions (Laist et al. 2001). Between 1999 and 2005, there were three reports of sei whales being struck by vessels along the U.S. Atlantic coast and Canada's Maritime Provinces (Cole et al. 2005b; Nelson et al. 2007b). Two of these ship strikes were reported as having resulted in death. One sei whale was killed in a collision with a vessel off the coast of Washington in 2003 (Waring et al. 2008). New rules for seasonal (June through December) slowing of vessel traffic in the Bay of Fundy to 10 knots and changing shipping lanes by less than one nautical mile to avoid the greatest concentrations of right whales are predicted to reduce sei whale ship strike mortality by 17%.

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

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

Humpback whale

Description of the species. Humpback whales are a cosmopolitan species that occur in the Atlantic, Indian, Pacific, and Southern oceans. Humpback whales migrate seasonally between warmer, tropical or sub-tropical waters in winter months (where they breed and give birth to calves, although feeding occasionally occurs) and cooler, temperate or sub-Arctic waters in summer months (where they feed; (Gendron and Urban 1993). In both regions, humpback whales tend to occupy shallow, coastal waters. However, migrations are undertaken through deep, pelagic waters (Winn and Reichley 1985).

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

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

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

North Pacific. Based on genetic and photo-identification studies, the NMFS currently recognizes four stocks, likely corresponding to populations, of humpback whales in the North Pacific Ocean: two in the eastern North Pacific, one in the central North Pacific, and one in the western Pacific (Hill and DeMaster 1998). However, there gene flow between them may exist. Humpback whales summer in coastal and inland waters from Point Conception, California, north to the Gulf of Alaska and the Bering Sea, and west along the Aleutian Islands to the Kamchatka Peninsula and into the Sea of Okhotsk (Johnson and Wolman 1984; Nemoto 1957; Tomilin 1967). These whales migrate to Hawaii, southern Japan, the Mariana Islands, and Mexico during winter. However, more northerly penetrations in Arctic waters occur on occasion (Hashagen et al. 2009). The central North Pacific population winters in the waters around Hawaii while the eastern North Pacific population (also called the California-Oregon-Washington-Mexico stock) winters along Central America and Mexico. However, Calambokidis et al. (1997) identified individuals from several populations wintering (and potentially breeding) in the areas of other populations, highlighting the potential fluidity of population structure. Herman (1979) presented extensive evidence that humpback whales associated with the main Hawaiian Islands immigrated there only in the past 200 years. Winn and Reichley (1985) identified genetic exchange between the humpback whales that winter off Hawaii and Mexico (with further mixing on feeding areas in Alaska) and suggested that humpback whales that winter in Hawaii may have emigrated from

Mexican wintering areas. A "population" of humpback whales winters in the South China Sea east through the Philippines, Ryukyu Retto, Ogasawara Gunto, Mariana Islands, and Marshall Islands, with occurrence in the Mariana Islands, at Guam, Rota, and Saipan from January-March (Darling and Mori 1993; Eldredge 1991; Eldredge 2003; Rice 1998a). During summer, whales from this population migrate to the Kuril Islands, Bering Sea, Aleutian Islands, Kodiak, Southeast Alaska, and British Columbia to feed (Angliss and Outlaw 2007b; Calambokidis 1997; Calambokidis et al. 2001).

Separate feeding groups of humpback whales are thought to inhabit western U.S. and Canadian waters, with the boundary between them located roughly at the U.S./Canadian border (Carretta et al. 2006). The southern feeding ground ranges between 32°-48°N, with limited interchange with areas north of Washington State (Calambokidis et al. 2004a; Calambokidis et al. 1996). Humpback whales feed along the coasts of Oregon and Washington from May-November, with peak numbers reported May-September, when they are the most commonly reported large cetacean in the region (Calambokidis et al. 2004a; Calambokidis et al. 2000; Dohl et al. 1983; Forney and Barlow 1998; Green et al. 1992). Off Washington State, humpback whales concentrate between Juan de Fuca Canyon and the outer edge of the shelf break in a region called "the Prairie," near Barkley and Nitnat canyons, in the Blanco upwelling zone, and near Swiftsure Bank (Calambokidis et al. 2004b). Humpback whales also tend to congregate near Heceta Bank off the coast of Oregon (Green et al. 1992). Additional data suggest that further subdivisions in feeding groups may exist, with up to six feeding groups present between Kamchatka and southern California (Witteveen et al. 2009).

Humpback whales primarily feed along the shelf break and continental slope (Green et al. 1992; Tynan et al. 2005). Although humpback whales were common in inland Washington State waters in the early 1900s, severe hunting throughout the eastern North Pacific has diminished their numbers and few recent inshore sighting have been made (Calambokidis and Steiger 1990; Pinnell and Sandilands 2004; Scheffer and Slipp 1948).

Similarly, humpback whales do not appear to frequent offshore waters in the region. Extensive aerial surveys conducted up to 550 km off the Oregonian and Washingtonian coasts, only one humpback whale was reported in offshore waters >200 m deep (Green et al. 1992). Encounter rates off Oregon/Washington during the summer were highest over the slope (2.16/1,000 individuals/km^2) followed by shelf waters (0.56 /1,000 individuals/km^2), with no sightings in offshore waters during the summer.

Southern Hemisphere. Eight proposed stocks, or populations, of humpback whales occur in waters off Antarctica (Figure 5). Individuals from these stocks winter and breed in separate areas and are known to return to the same areas. However, the degree (if any) of gene flow (i.e., adult individuals wintering in different breeding locations) is uncertain. Based upon recent satellite telemetry, a revision of stocks A and G may be warranted to reflect stock movements within and between feeding areas separated east of 50º W (Dalla Rosa et al. 2008). A separate population of humpback whales appears to reside in the Arabian Sea in the Indian Ocean off the coasts of Oman, Pakistan, and India and movements of this group are poorly known (Mikhalev 1997; Rasmussen et al. 2007). Areas of the Mozambique Channel appear to be significant calving and wintering areas for humpback whales (Kiszka et al. 2010a). In addition to being a breeding area, the west coast of South Africa also appears to serve as a

foraging ground due to upwelling of the Benguela Current (Barendse et al. 2010). In addition, females appear in this area in large numbers well before their male counterparts, frequently accompanied by calves (Barendse et al. 2010).

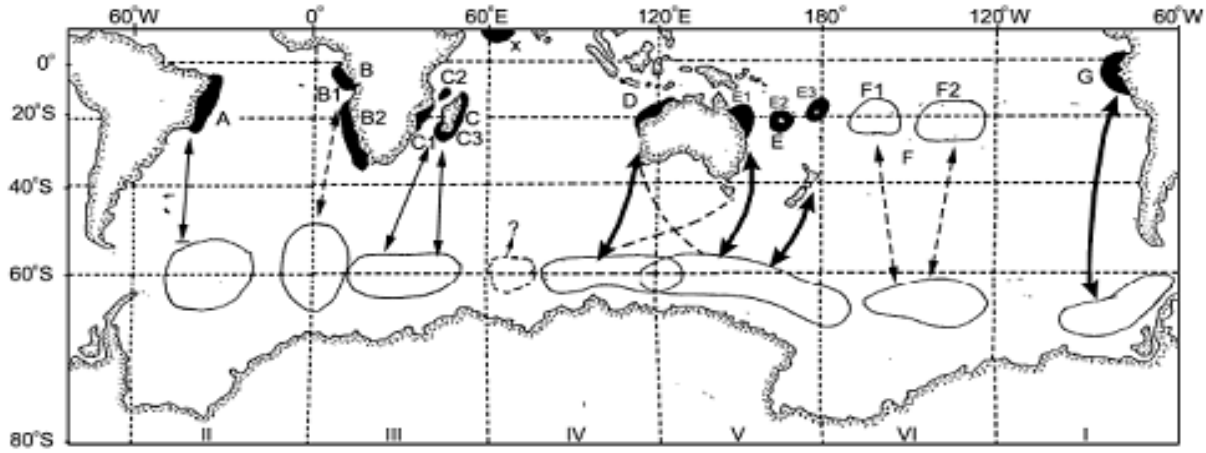

Figure 5. Southern Hemisphere humpback stocks (populations) (IWC 2005).

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

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

Diving. In Hawaiian waters, humpback whales remain almost exclusively within the 1,800 m isobath and usually within waters depths of less than 182 m. Maximum diving depths are approximately 170 m (but usually <60 m), with a very deep dive (240 m) recorded off Bermuda (Hamilton et al. 1997). Dives can last for up to 21 min, although feeding dives ranged from 2.1-5.1 min in the north Atlantic (Dolphin 1987). In southeast Alaska, average dive times were 2.8 min for feeding whales, 3.0 min for non-feeding whales, and 4.3 min for resting whales (Dolphin 1987). In the Gulf of California, humpback whale dive durations averaged 3.5 min (Strong

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

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

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

Status and trends. Humpback whales were originally listed as endangered in 1970 (35 FR 18319), and this status remains under the ESA. (Winn and Reichley 1985) argued that the global humpback whale population consisted of at least 150,000 whales in the early 1900s, mostly in the Southern Ocean. In 1987, the global population of humpback whales was estimated at about 10,000 (NMFS 1987). Although this estimate is outdated, it appears that humpback whale numbers are increasing. Table 5 provides estimates of historic and current abundance for ocean regions.

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

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

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

North Atlantic. The best available estimate of North Atlantic abundance comes from 1992-1993 mark-recapture data, which generated an estimate of 11,570 humpback whales (Stevick et al. 2003a). Historical estimates have ranged from 40,000-250,000 individuals and significant disagreement exists on how many humpbacks whales inhabited the North Atlantic prior to whaling (Smith and G.Pike 2009). Estimates of animals in Caribbean breeding grounds exceed 2,000 individuals (Balcomb III and Nichols 1982). Several researchers report an increasing trend in abundance for the North Atlantic population, which is supported by increased sightings within the Gulf of Maine feeding aggregation (Barlow 1997a; Katona and Beard 1990; Smith et al. 1999; Waring et al. 2001). The rate of increase varies from 3.2-9.4%, with rates of increase slowing over the past two decades (Barlow 1997a; Katona and Beard 1990; Stevick et al. 2003a). If the North Atlantic population has grown according to the estimated instantaneous rate of increase (r = 0.0311), this would lead to an estimated 18,400 individual whales in 2008 (Stevick et al. 2003a). Pike et al. (2009a) suggested that the eastern and northeastern waters off Iceland are areas of significant humpback utilization for feeding, estimating nearly 5,000 whales in 2001 and proposing an annual growth rate of 12% for the area. The authors went so far as to suggest that humpback whales in the area had probably recovered from whaling.

North Pacific. The pre-exploitation population size may have been as many as 15,000 humpback whales, and current estimates are 6,000-8,000 whales (Calambokidis et al. 1997; Rice 1978a). It is estimated that 15,000 humpback whales resided in the North Pacific in 1905 (Rice

1978a). However, from 1905 to 1965, nearly 28,000 humpback whales were harvested in whaling operations, reducing the number of all North Pacific humpback whale to roughly 1,000 (Perry et al. 1999). Estimates have risen over time from 1,407-2,100 in the 1980s to 6,010 in 1997 (Baker 1985; Baker and Herman 1987; Calambokidis et al. 1997; Darling and Morowitz 1986). Because estimates vary by methodology, they are not directly comparable and it is not clear which of these estimates is more accurate or if the change from 1,407 to 6,010 is the result of a real increase or an artifact of model assumptions. Tentative estimates of the eastern North Pacific stock suggest an increase of 6-7% annually, but fluctuations have included negative growth in the recent past (Angliss and Outlaw 2005a). However, based upon surveys between 2004 and 2006, Calambokidis et al. (2008) estimated that the number of humpback whales in the North Pacific consisted of about 18,300 whales, not counting calves. Almost half of these whales likely occur in wintering areas around the Hawaiian Islands.

Southern Hemisphere. The IWC recently compiled population data on humpback whales in the Southern Hemisphere. Approximately 42,000 Southern Hemisphere humpbacks can be found south of 60° S during the austral summer feeding season (IWC 2007). However, humpback whales in this region experienced severe whaling pressure. Based upon whaling logs, particularly by Soviet vessels, at least 75,542 humpback whales were harvested from Antarctic waters from 1946 through 1973, largely from management areas IV, V, and VI (Clapham et al. 2009). One-third of these catches occurred from 1959-1961 in Area V. These numbers support Southern Hemisphere humpbacks being well below their carrying capacities (Clapham et al. 2009). Recent surveys off the Brazilian breeding grounds suggests a populations of 6,404 individuals in this area (Andriolo et al. 2010).

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

Parasites and biotoxins from red-tide blooms are other potential causes of mortality (Perry et al. 1999). The occurrence of the nematode *Crassicauda boopis* appears to increase the potential for kidney failure in humpback whales and may be preventing some populations from recovering (Lambertsen 1992). Studies of 14 humpback whales that stranded along Cape Cod between November 1987 and January 1988 indicate they apparently died from a toxin produced by dinoflagellates during this period.

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

Humpback whales are also killed or injured during interactions with commercial fishing gear. Like fin whales, humpback whales have been entangled by fishing gear off Newfoundland and Labrador, Canada. A total of 595 humpback whales were reported captured in coastal fisheries

in those two provinces between 1969 and 1990, of which 94 died (Lien 1994; Perkins and Beamish 1979). Along the Atlantic coast of the U.S. and the Maritime Provinces of Canada, there were 160 reports of humpback whales being entangled in fishing gear between 1999 and 2005 (Cole et al. 2005b; Nelson et al. 2007b). Of these, 95 entangled humpback whales were confirmed, with 11 whales sustaining injuries and nine dying of their wounds. Several humpback whales are also known to have become entangled in the North Pacific (Angliss and Outlaw 2007b; Hill et al. 1997).

More humpback whales are killed in collisions with ships than any other whale species except fin whales (Jensen and Silber 2003a). Along the Pacific coast, a humpback whale is known to be killed about every other year by ship strikes (Barlow et al. 1997). Of 123 humpback whales that stranded along the Atlantic coast of the U.S. between 1975 and 1996, 10 (8.1%) showed evidence of collisions with ships (Laist et al. 2001). Between 1999 and 2005, there were 18 reports of humpback whales being struck by vessels along the Atlantic coast of the U.S. and the Maritime Provinces of Canada (Cole et al. 2005b; Nelson et al. 2007b). Of these reports, 13 were confirmed as ship strikes and in seven cases, ship strike was determined to be the cause of death. In the Bay of Fundy, recommendations for slower vessel speeds to avoid right whale ship strike appear to be largely ignored (Vanderlaan et al. 2008). However, new rules for seasonal (June through December) slowing of vessel traffic to 10 knots and changing shipping lanes by less than one nautical mile to avoid the greatest concentrations of right whales are expected to reduce the chance of humpback whales being hit by ships by 9%.

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

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

Bowhead whale

Description of the species. Currently, five bowhead whale stocks have been identified: Sea of Okhotsk, Davis Strait, Hudson Bay, offshore waters of Spitsbergen, and the western Arctic, with only the latter occurring in U.S. waters, and most stocks consist of a few dozens to hundreds of individuals (Ivashchenko and Clapham 2010; IWC 1992a; NMFS 2006i). Genetically, significant genetic differentiation exists between these areas (Givens et al. 2010; Ivashchenko and Clapham 2010). However, genetic analyses have thus far not clearly identified differences, particularly between Atlantic stocks, although some differentiation in haplotypes appears to exist between Hudson Bay and Davis Strait individuals in some areas (but not in all areas) (Bachmann

et al. 2010; Heide-Jorgensen and Postma 2006; Postma and Cosens 2006). Genetic differentiation appears to be high within the western Arctic stock, which likely represents a single population (Givens et al. 2010).

Distribution. Bowhead whales only occur at high latitudes in the northern hemisphere and have a disjunctive circumpolar distribution (Reeves 1980). Bowhead whales are found in the western Arctic (Bering, Chukchi, and Beaufort Seas), the Canadian Arctic and West Greenland (Baffin Bay, Davis Strait, and Hudson Bay), the Okhotsk Sea (eastern Russia), and the Northeast Atlantic from Spitzbergen westward to eastern Greenland. In the Chukchi Sea, bowheads are found in all months of the year (mainly west and southwest of Point Barrow) and distribution does not appear linked to changes in sea ice cover (Clarke and Ferguson. 2010b). Bowheads inhabiting the Okhotsk Sea appear to reside there year-round (Ivashchenko and Clapham 2010). Historically, bowhead whale range has extended into the eastern Atlantic, in which basin it is estimated that 52,500 individuals once lived (Allen et al. 2006).

Movement and habitat. The majority of the western Arctic stock migrates annually from wintering areas (November to March) in the northern Bering Sea, through the Chukchi in spring (March through June), to the Beaufort Sea where they spend much of the summer (mid-May through November) before returning again to the Bering Sea in fall. In the Chukchi Sea, bowheads are generally found in waters between 50 and 200 m deep (Clarke and Ferguson. 2010b). However, individuals in the Beaufort Sea appear to strongly favor shallower areas less than 50 m and preferably shallower than 20 m (Clarke and Ferguson. 2010a). Feeding appears to preferentially occur in 154-157º longitude in the Beaufort Sea (Clarke and Ferguson. 2010a). During their migrations north, they are forced between land and pack ice around Point Barrow, Alaska. They spend most of the summer in relatively ice-free waters of the Beaufort Sea, but they are associated with sea ice the rest of the year (Moore and Reeves 1993). During their autumn migration, bowhead whales preferentially select nearshore shelf waters, except if there are heavy ice conditions, in which case they select slope habitat. Not all bowhead whales follow this migration and some over-summer in the Bering and Chukchi Seas.

Growth and reproduction. Reproductive activities for bowhead whales occur throughout the year, but conception takes place in late winter or early spring. Some whales may be unable to conceive, as there is evidence of pseudohermaphroditism in a relatively high percentage (two of 76 whales sampled) of male bowhead whales (Philo et al. 1992). Gestation lasts 12 to 16 months and the calving interval is between 3.5 and seven years (Nerini et al. 1984; Tarpley et al. 1995). Bowhead whales take approximately two decades to become sexually mature, when they reach approximately 40 to 46 feet in length (IWC 2004a; Nerini et al. 1984; Schell and Saupe 1993; Schell et al. 1989). Disko Bay, Canada has been proposed as a breeding site for bowheads in the Baffin Bay stock and Foxe Bay has been proposed as a calf-rearing site (Heide-Jorgensen et al. 2010b).

Feeding. Bowhead whales in the North Pacific feed on euphausiids and copepods, which make up most of their diets (Heide-Jorgensen et al. 2010a; Lowry 1993).

Diving. Bowhead diving behavior is situational (Stewart 2002). Calves dive for very short periods and their mothers tend to dive less frequently and for shorter durations. Feeding dives tend to last from three to 12 minutes and may extend to the relatively shallow bottom in the

Beaufort Sea. "Sounding" dives average between 7 and 14 minutes. When individuals migrate through pack ice, dives tend to become longer and deeper, presumably to navigate through areas where breathing holes may not be accessible. However, when harassed by whalers, bowheads are known to dive for as long as 80 minutes.

Status and trends. Bowhead whales were originally listed as endangered in 1970 (35 FR 18319), and this status remained since the inception of the ESA in 1973. Bowhead whale abundance prior to commercial whaling in the western Arctic has been estimated at 10,400 to 23,000 (Woodby and Botkin 1993). At the end of commercial whaling the species had declined to between 1,000 and 3,000 bowhead whales in the western Arctic. The current minimum population estimate is 9,472 whales, and in 2001 the population was estimated at 10,545 individuals (Angliss and Outlaw 2008). Also in 2001, 121 calves were counted, which is the most calves recorded in a single year. The population has been increasing at approximately 3.1% from 1978 to 1993 and more recently by about 3.5% annually (Angliss and Outlaw 2008).

This upward population trend is consistent with impressions of local hunters and western Arctic recovery may warrant delisting in the future (Gerber et al. 2007; Noongwook et al. 2007). It is also estimated that 1,229 individuals reside in the Spitsbergen stock, which also exceeds prior abundance estimates and sightings are occurring on a more regular basis (Gilg and Born 2005; Heide-Jorgensen et al. 2007). In 2009, a calf was spotted off northeast Greenland; the first observed in the Spitsbergen stock in 18 years (Boertmann and Nielsen. 2010). Hansen et al. (2010) estimated 1,105 individuals in Isabella Bay, Canada in September 2009. The eastern Canada-western Greenland stock appears to be increasing robustly based upon age at sexual maturity and calving interval data (Koski et al. 2010).

Natural threats. Little is known of diseases and natural death in the western Arctic bowhead whale population, but the mortality rate is thought to be low (Koski et al. 1993). Bowhead whales have been subjects of killer whale attacks and, because of their robust size and slow swimming speed, tend to form small groups and fight killer whales when confronted and may cause killer whale mortality with their flukes (Ford and Reeves 2008). Individuals have been known to be trapped by sea ice for extended periods, which may pose a lethal threat.

Anthropogenic threats. Bowhead whales began declining precipitously with directed whaling efforts in the Bering Sea between 1850 and 1870, when an estimated 60% of individuals were harvested (Braham 1984b). Harvests declined after 1870, although whaling efforts continued, including illegal Soviet whaling (Ivashchenko and Clapham 2010). Subsistence harvests continue at present, with 31 of 38 whales struck by Alaskan native harpoons killed and landed in 2009, which is roughly similar to annual landings over the past decade (Suydam et al. 2004; Suydam et al. 2009; Suydam et al. 2010; Suydam et al. 2005; Suydam et al. 2006; Suydam et al. 2003; Suydam and George. 2004; Suydam et al. 2002).

Present threats to bowhead whales include interactions with crab pots, nets, and ship propellers at low levels. Between 1978 and 2004, eight bowheads were observed entangled and five had propeller scars (NMFS 2006i). These bowheads likely became entangled as a result of "skimming" prey at the water's surface and becoming entangled with debris. More significant are the number of bowhead whales taken by native tribes from the western Arctic stock: 14 to 72 individuals, or 0.1 to 0.5% of the stock population annually. Under this system, 832 individuals are known to have been taken from 1974 to 2003. However, these hunts are closely monitored

and accessed for negative impacts on population number and structure and serve to maintain tribal culture. Individuals are known to have been taken by native tribes in Canada and Russia, although in extremely low numbers. Another potential threat is the documented reduction in sea ice, weather, or temperature conditions that has resulted from global warming (Tynan and DeMaster 1997). It is unknown what effects these large scale changes may have (NMFS 2006i).

Several contaminants have been isolated from bowhead whale tissues in low concentrations, including organochlorines, mercury, lead, arsenic, zinc, copper, cadmium, selenium, and silver (Dehn et al. 2006; O'Hara et al. 2006; Rosa et al. 2007b). Rosa et al. (2008) measured metal concentrations in the liver that included zinc (6.99 to 135.11 mg/kg wet weight), copper (1.09 to 203.81 mg/kg), cadmium (0.003 to 50.91 mg/kg), selenium (0.06 to 3.77 mg/kg), silver (0.05 to 2.37 mg/kg), and mercury (0.001 to 0.47 mg/kg). These same metals in kidney are generally lower, but present; zinc (9.07 to 56.31 mg/kg wet weight), copper (0.76 to 7.94 mg/kg), cadmium (0.01 to 64.0 mg/kg), selenium (0.23 to 3.21 mg/kg), silver (0.01 to 0.06 mg/kg), and mercury (0.001 to 0.14 mg/kg). Thickening of the Bowman's capsules and fibrous tissue formations are associated with cadmium accumulation in the kidney. These changes may reduce kidney function, although bowheads seem to be able to withstand significant kidney pathology (Parrish et al. 2008). Bioaccumulation of these metals occurs with age, but differences between sexes have not been observed in metal concentration (Parrish et al. 2008). These concentrations are lower than in other studied cetaceans due to the lower level at which bowhead whales feed in the overall food chain (Dehn et al. 2006; Parrish et al. 2008). Hormonal concentrations suggest that contaminants are not presently a significant hindrance for bowhead whales (Rosa et al. 2007a). However, the development of Arctic regions for oil and gas can increase contaminant loads in the environment, prey species, and protected species such as bowhead whales. Organochlorine levels are also believed to accumulate in arctic regions (Tanabe et al. 1994), leading to concern over the potential bioaccumulation of these toxins in bowhead whales due to global sources.

Bowhead whales have also been shown to vacate areas in which drilling and seismic survey operations occur, apparently in response to sound (Davies 1997; Miller et al. 1999b; Richardson 1995; Richardson and Malme 1993; Schick and Urban 2000). It is possible that migratory routes have already shifted in response to anthropogenic sound (Richardson et al. 2004a).

Critical habitat. NMFS has not designated critical habitat for bowhead whales.

North Pacific right whale

Description of the species. Many basic life history parameters of North Pacific right whales are unknown. All North Pacific right whales constitute a single population.

Distribution. Very little is known of the distribution of right whales in the North Pacific and very few of these animals have been seen in the past 20 years. Historical whaling records indicate that right whales ranged across the North Pacific north of 30° N latitude and occasionally as far south as 20° N, with a bimodal distribution longitudinally favoring the eastern and western North Pacific and occurring infrequently in the central North Pacific (Gregr and Coyle. 2009; Josephson et al. 2008a; Maury 1853; Scarff 1986; Scarff 1991; Townsend 1935a). North Pacific right whales summered in the North Pacific and southern Bering Sea from April or May to September, with a peak in sightings in coastal waters of Alaska in June and July (Klumov 1962; Maury 1852; Omura 1958; Omura et al. 1969a; Townsend 1935a). North Pacific right

whale summer range extended north of the Bering Strait (Omura et al. 1969a). However, they were particularly abundant in the Gulf of Alaska from 145° to 151°W, and apparently concentrated in the Gulf of Alaska, especially south of Kodiak Islands and in the eastern Aleutian Islands and southern Bering Sea waters (Berzin and Rovnin 1966; Braham and Rice 1984).

Current information on the seasonal distribution of right whales is spotty. In the eastern North Pacific, this includes sightings over the middle shelf of the Bering Sea, Bristol Bay, Aleutian and Pribilof Islands (Goddard and Rugh 1998; Hill and DeMaster 1998; Perryman et al. 1999; Wade et al. 2006b; Waite et al. 2003). Some more southerly records also record occurrence along Hawaii, California, Washington, and British Columbia (Herman et al. 1980; Scarff 1986). However, records from Mexico and California may suggest historical wintering grounds in offshore southern North Pacific latitudes (Brownell et al. 2001a; Gregr and Coyle. 2009).

Growth and reproduction. While no reproductive data are known for the North Pacific, studies of North Atlantic right whales suggest calving intervals of two to seven years and growth rates that are likely dependent on feeding success (Best et al. 2001; Burnell 2001; Cooke et al. 2001; Kenney 2002; Knowlton et al. 1994; Reynolds et al. 2002). It is presumed that right whales calve during mid-winter (Clapham et al. 2004a). Western North Pacific sightings have been recorded along Japan, the Yellow Sea, and Sea of Japan (Best et al. 2001; Brownell et al. 2001b, areas that are speculated to be important breeding and calving areas).

Lifespan. Lifespans of up to 70 years can be expected based upon North Atlantic right whale data.

Feeding. Stomach contents from North Pacific right whales indicate copepods and, to a lesser extent, euphausiid crustaceans are the whales' primary prey (Omura et al. 1969b). Their diet is likely more varied than North Atlantic right whales, likely due to the multiple blooms of different prey available in the North Pacific from January through August (Gregr and Coyle. 2009). Based upon trends in prey blooms, it is predicted that North Pacific right whales may shift from feeding offshore to over the shelf edge during late summer and fall (Gregr and Coyle. 2009). North Pacific right whales, due to the larger size of North Pacific copepods, have been proposed to be capable to exploit younger age classes of prey as well as a greater variety of species. Also as a result, they may require prey densities that are one-half to one-third those of North Atlantic right whales (Gregr and Coyle. 2009). Right whales feed by continuously filtering prey through their baleen while moving, mouth agape, through patches of planktonic crustaceans. Right whales are believed to rely on a combination of experience, matrilinear learning, and sensing of oceanographic conditions to locate prey concentrations in the open ocean (Gregr and Coyle. 2009; Kenney 2001).

Habitat. Habitat preference data are sparse for North Pacific right whales as well. Sightings have been made with greater regularity in the western North Pacific, notably in the Okhotsk Sea, Kuril Islands, and adjacent areas (Brownell et al. 2001b). In the western North Pacific, feeding areas occur in the Okhotsk Sea and adjacent waters along the coasts of Kamchatka and the Kuril Islands (IWC 2001).

Historical concentrations of sightings in the Bering Sea together with some recent sightings indicate that this region, together with the Gulf of Alaska, may represent an important summer habitat for eastern North Pacific right whales (Brownell et al. 2001b; Clapham et al. 2004a;

Goddard and Rugh 1998; Scarff 1986; Shelden et al. 2005a). Few sighting data are available from the eastern North Pacific, with a single sighting of 17 individuals in the southeast Bering Sea being by far the greatest known occurrence (Wade et al. 2006a). Some further sightings have occurred in the northern Gulf of Alaska (Wade et al. 2006a). Recent eastern sightings tend to occur over the continental shelf, although acoustic monitoring has identified whales over abyssal waters (Mellinger et al. 2004). It has been suggested that North Pacific right whales have shifted their preferred habitat as a result of reduced population numbers, with oceanic habitat taking on a far smaller component compared to shelf and slope waters (Shelden et al. 2005b).

Migration and movement. Historical sighting and catch records provide the only information on possible migration patterns for North Pacific right whales (Omura 1958; Omura et al. 1969a; Scarff 1986). During summer, whales have been found in the Gulf of Alaska, along both coasts of the Kamchatka Peninsula, the Kuril Islands, the Aleutian Islands, the southeastern Bering Sea, and in the Okhotsk Sea. Fall and spring distribution was the most widely dispersed, with whales occurring in mid-ocean waters and extending from the Sea of Japan to the eastern Bering Sea. In winter, right whales have been found in the Ryukyu Islands (south of Kyushu, Japan), the Bonin Islands, the Yellow Sea, and the Sea of Japan. Whalers never reported winter calving areas in the North Pacific and where calving occurs remains unknown (Clapham et al. 2004a; Gregr and Coyle. 2009; Scarff 1986). North Pacific right whales probably migrate north from lower latitudes in spring and may occur throughout the North Pacific from May through August north of 40º N from marginal seas to the Gulf of Alaska and Bering Sea, although absence from the central North Pacific has been argued due to inconsistencies in whaling records (Clapham et al. 2004b; Josephson et al. 2008b). This follows generalized patterns of migration from high-latitude feeding grounds in summer to more temperate, possibly offshore waters, during winter (Braham and Rice 1984; Clapham et al. 2004a; Scarff 1986).

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

Very little is known about right whales in the eastern North Pacific, which were severely depleted by commercial whaling in the 1800s (Brownell et al. 2001b). At least 11,500 individuals were taken by American whalers in the early- to mid-19th century, but harvesting continued into the 20th century (Best 1987). Illegal Soviet whaling took 372 individuals between 1963 and 1967 (Brownell et al. 2001a). In the last several decades there have been markedly fewer sightings due to a drastic reduction in number, caused by illegal Soviet whaling in the 1960s (Doroshenko 2000). Previous estimates of the size of the right whale population in the Pacific Ocean range from a low of 100-200 (Braham and Rice 1984) to a high of 220-500 (Berzin and Yablokov 1978). The current population size of right whales in the North Pacific is likely fewer than 1,000 animals (NMFS 2006g).

Abundance estimates and other vital rate indices in both the eastern and western North Pacific

are not well established. Where such estimates exist, they have very wide confidence limits. Previous estimates of the size of the right whale population in the Pacific Ocean range from a low of 100-200 to a high of 220-500 (Berzin and Yablokov 1978; Braham and Rice 1984). Although Hill and DeMaster (1998) argued that it is not possible to reliably estimate the population size or trends of right whales in the North Pacific, Reeves et al. (2003a) concluded that North Pacific right whales in the eastern Pacific Ocean exist as a small population of individuals while the western population of right whales probably consists of several hundred animals, although Clapham et al. (2005) placed this population at likely under 100 individuals. Brownell et al. (2001b) reviewed sighting records and also estimated that the abundance of right whales in the western North Pacific was likely in the low hundreds.

Scientists participating in a recent study utilizing acoustic detection and satellite tracking identified 17 right whales (10 males and 7 females) in the Bearing Sea, which is almost threefold the number seen in any previous year in the last four decades (Wade et al. 2006b). These sightings increased the number of individual North Pacific right whales identified in the genetic catalog for the eastern Bering Sea to 23. Amidst the uncertainty of the eastern North Pacific right whale's future, the discovery of females and calves gives hope that this endangered population may still possess the capacity to recover (Wade et al. 2006b). Available age composition of the North Pacific right whale population indicates a most individuals are adults of adults (Kenney 2002). Length measurements for two whales observed off California suggest at least one of these whales was not yet sexually mature and two calves have been observed in the Bering Sea (Carretta et al. 1994; Wade et al. 2006b). However, to date, there is no evidence of reproductive success (i.e., young reared to independence) in the eastern North Pacific. No data are available for the western North Pacific.

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

Anthropogenic threats. Whaling for North Pacific right whales was discontinued in 1966 with the IWC whaling moratorium. However, North Pacific right whales remain at considerable risk of extinction. These include but are not limited to the following: (1) life history characteristics such as slow growth rate, long calving intervals, and longevity; (2) distorted age structure of the population and reduced reproductive success; (3) strong depensatory or Allee effects; (4) habitat specificity or site fidelity; and (5) habitat sensitivity. However, the proximity of the other known right whale habitats to shipping lanes (e.g. Unimak Pass) suggests that collisions with vessels may also represent a threat to North Pacific right whales (Elvin and Hogart 2008).

Climate change may have a dramatic affect on survival of North Pacific right whales. Right whale life history characteristics make them very slow to adapt to rapid changes in their habitat (see Reynolds et al. 2002). They are also feeding specialists that require exceptionally high densities of their prey (see Baumgartner et al. 2003; Baumgartner and Mate 2003a). Zooplankton abundance and density in the Bering Sea has been shown to be highly variable, affected by climate, weather, and ocean processes and in particular ice extent (Baier and Napp 2003; Napp and G.L. Hunt 2001). The largest concentrations of copepods occurred in years with the greatest

southern extent of sea ice (Baier and Napp 2003). It is possible that changes in ice extent, density and persistence may alter the dynamics of the Bering Sea shelf zooplankton community and in turn affect the foraging behavior and success of right whales. No data are available for the western North Pacific.

Critical habitat. In July 2006, NMFS designated two areas as critical habitat for right whales in the North Pacific (71 FR 38277). The areas encompass about 36,750 square miles of marine habitat, which include feeding areas within the Gulf of Alaska and the Bering Sea that support the species. The primary constituent element to this critical habitat is the presence of large copepods and oceanographic factors that concentrate North Pacific right whale prey. At present, this PCE has not been significantly degraded due to human activity. However, significant concern has been voiced regarding the impact that oceanic contamination of pollutants may have on the food chain and consequent bioaccumulation of toxins by marine predators. Changes due to global warming have also been raised as a concern that could affect the distribution or abundance of copepod prey for several marine mammals, including right whales.

Sperm whale

Description of the species. Sperm whales are distributed in all of the world's oceans, from equatorial to polar waters, and are highly migratory. Mature males range between 70º N in the North Atlantic and 70º S in the Southern Ocean (Perry et al. 1999; Reeves and Whitehead 1997), whereas mature females and immature individuals of both sexes are seldom found higher than 50º N or S (Reeves and Whitehead 1997). In winter, sperm whales migrate closer to equatorial waters (Kasuya and Miyashita 1988; Waring et al. 1993) where adult males join them to breed.
Stock designations. There is no clear understanding of the global population structure of sperm whales (Dufault et al. 1999). Recent ocean-wide genetic studies indicate low, but statistically significant, genetic diversity and no clear geographic structure, but strong differentiation between social groups (Lyrholm and Gyllensten 1998; Lyrholm et al. 1996; Lyrholm et al. 1999). The IWC currently recognizes four sperm whale stocks: North Atlantic, North Pacific, northern Indian Ocean, and Southern Hemisphere (Dufault et al. 1999; Reeves and Whitehead 1997). The NMFS recognizes six stocks under the MMPA- three in the Atlantic/Gulf of Mexico and three in the Pacific (Alaska, California-Oregon-Washington, and Hawaii; (Perry et al. 1999; Waring et al. 2004b). Genetic studies indicate that movements of both sexes through expanses of ocean basins are common, and that males, but not females, often breed in different ocean basins than the ones in which they were born (Whitehead 2003). Sperm whale populations appear to be structured socially, at the level of the clan, rather than geographically (Whitehead 2003; Whitehead et al. 2008).

North Atlantic. In the western North Atlantic, sperm whales range from Greenland south into the Gulf of Mexico and the Caribbean, where they are common, especially in deep basins north of the continental shelf (Romero et al. 2001; Wardle et al. 2001). The northern distributional limit of female/immature pods is probably around Georges Bank or the Nova Scotian shelf (Whitehead et al. 1991). Seasonal aerial surveys confirm that sperm whales are present in the northern Gulf of Mexico in all seasons (Hansen et al. 1996; Mullin et al. 1994). Sperm whales distribution follows a distinct seasonal cycle, concentrating east-northeast of Cape Hatteras in winter and shifting northward in spring when whales are found throughout the Mid-Atlantic Bight. Distribution extends further northward to areas north of Georges Bank and the Northeast Channel region in summer and then south of New England in fall, back to the Mid-

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

North Pacific. Sperm whales are found throughout the North Pacific and are distributed broadly in tropical and temperate waters to the Bering Sea as far north as Cape Navarin in summer, and occur south of 40° N in winter (Gosho et al. 1984; Miyashita et al. 1995 as cited in Carretta et al. 2005; Rice 1974). Sperm whales are found year-round in Californian and Hawaiian waters (Barlow 1995; Dohl et al. 1983; Forney et al. 1995; Lee 1993; Mobley Jr . et al. 2000; Rice 1960; Shallenberger 1981b), but they reach peak abundance from April-mid-June and from the end of August-mid-November (Rice 1974). They are seen in every season except winter (December-February) in Washington and Oregon (Green et al. 1992). Summer/fall surveys in the eastern tropical Pacific (Wade and Gerrodette 1993) show that although sperm whales are widely distributed in the tropics, their relative abundance tapers off markedly towards the middle of the tropical Pacific and northward towards the tip of Baja California (Carretta et al. 2006).

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

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

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

Gaskin (1973) proposed a northward population shift of sperm whales off New Zealand in the

austral autumn based on reduction of available food species and probable temperature tolerances of calves.

Habitat. Sperm whales have a strong preference for waters deeper than 1,000 m (Reeves and Whitehead 1997; Watkins 1977), although Berzin (1971) reported that they are restricted to waters deeper than 300 m. While deep water is their typical habitat, sperm whales are rarely found in waters less than 300 m in depth (Clarke 1956; Rice 1989a). Sperm whales have been observed near Long Island, New York, in water between 40-55 m deep (Scott and Sadove 1997). When they are found relatively close to shore, sperm whales are usually associated with sharp increases in topography where upwelling occurs and biological production is high, implying the presence of a good food supply (Clarke 1956). Such areas include oceanic islands and along the outer continental shelf.

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

Local information is inconsistent regarding sperm whale tendencies. Gregr and Trites (2001) reported that female sperm whales off British Columbia were relatively unaffected by the surrounding oceanography. However, Tynan et al. (2005) reported increased sperm whales densities with strong turbulence associated topographic features along the continental slope near Heceta Bank. Two noteworthy strandings in the region include an infamous incident (well publicized by the media) of attempts to dispose of a decomposed sperm whale carcass on an Oregon beach by using explosives. In addition, a mass (Tynan et al. 2005) stranding of 47 individuals in Oregon occurred during June 1979 (Norman et al. 2004; Rice et al. 1986).

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

Sperm whale age distribution is unknown, but sperm whales are believed to live at least 60 years

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

Stable, long-term associations among females form the core of sperm whale societies (Christal et al. 1998). Up to about a dozen females usually live in such groups, accompanied by their female and young male offspring. Young individuals are subject to alloparental care by members of either sex and may be suckled by non-maternal individuals (Gero et al. 2009). Group sizes may be smaller overall in the Caribbean Sea (6-12 individuals) versus the Pacific (25-30 individuals) (Jaquet and Gendron 2009; Jaquet and Gendron. 2009). Groups may be stable for long periods, such as for 80 days in the Gulf of California (shorter in other areas) (Jaquet and Gendron 2009). Males start leaving these family groups at about 6 years of age, after which they live in "bachelor schools," but this may occur more than a decade later (Pinela et al. 2009). The cohesion among males within a bachelor school declines with age. During their breeding prime and old age, male sperm whales are essentially solitary (Christal and Whitehead 1997).

Diving. Sperm whales are probably the deepest and longest diving mammalian species, with dives to 3 km down and durations in excess of 2 hours (Clarke 1976; Watkins et al. 1993b; Watkins et al. 1985). However, dives are generally shorter (25- 45 min) and shallower (400-1,000 m). Dives are separated by 8-11 min rests at the surface (Gordon 1987; Jochens et al. 2006; Papastavrou et al. 1989; Watwood et al. 2006; Würsig et al. 2000b). Sperm whales typically travel ~3 km horizontally and 0.5 km vertically during a foraging dive (Whitehead 2003). Differences in night and day diving patterns are not known for this species, but, like most diving air-breathers for which there are data (rorquals, fur seals, and chinstrap penguins), sperm whales probably make relatively shallow dives at night when prey are closer to the surface.

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

Vocalization and hearing. Sound production and reception by sperm whales are better understood than in most cetaceans. Sperm whales produce broad-band clicks in the frequency range of 100 Hz to 20 kHz that can be extremely loud for a biological source (200-236 dB re 1μPa), although lower source level energy has been suggested at around 171 dB re 1 μPa (Goold and Jones 1995; Møhl et al. 2003; Weilgart and Whitehead 1993; Weilgart and Whitehead 1997). Most of the energy in sperm whale clicks is concentrated at around 2-4 kHz and 10-16 kHz (Goold and Jones 1995; NMFS 2006d; Weilgart and Whitehead 1993). The highly asymmetric

head anatomy of sperm whales is likely an adaptation to produce the unique clicks recorded from these animals (Cranford 1992; Norris and Harvey 1972; Norris and Harvey. 1972). These long, repeated clicks are associated with feeding and echolocation (Goold and Jones 1995; Weilgart and Whitehead 1993; Weilgart and Whitehead 1997). However, clicks are also used in short patterns (codas) during social behavior and intragroup interactions (Weilgart and Whitehead 1993). They may also aid in intra-specific communication. Another class of sound, "squeals", are produced with frequencies of 100 Hz to 20 kHz (e.g., Weir et al. 2007).

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

Status and trends. Sperm whales were originally listed as endangered in 1970 (35 FR 18319), and this status remained with the inception of the ESA in 1973. Although population structure of sperm whales is unknown, several studies and estimates of abundance are available. Table 5 contains historic and current estimates of sperm whales by region. Sperm whale populations probably are undergoing the dynamics of small population sizes, which is a threat in and of itself. In particular, the loss of sperm whales to directed Soviet whaling likely inhibits recovery due to the loss of adult females and their calves, leaving sizeable gaps in demographic and age structuring (Whitehead 2003).

North Atlantic. 190,000 sperm whales were estimated to have been in the entire North Atlantic, but CPUE data from which this estimate is derived are unreliable according to the IWC (Perry et al. 1999). The total number of sperm whales in the western North Atlantic is unknown (Waring et al. 2008). The best available current abundance estimate for western North Atlantic sperm whales is 4,804 based on 2004 data. The best available estimate for Northern Gulf of Mexico sperm whales is 1,665, based on 2003-2004 data, which are insufficient data to determine population trends (Waring et al. 2008). Sperm whale were widely harvested, from the northeastern Caribbean (Romero et al. 2001) and the Gulf of Mexico where sperm whale fishery operated during the late 1700s to the early 1900s (NMFS 2006e; Townsend 1935b).

North Pacific. There are approximately 76,803 sperm whales in the eastern tropical Pacific, eastern North Pacific, Hawaii, and western North Pacific (Whitehead 2002). Minimum estimates in the eastern North Pacific are 1,719 individuals and 5,531 in the Hawaiian Islands (Carretta et al. 2007c). The tropical Pacific is home to approximately 26,053 sperm whales and the western North Pacific has approximately 29,674 (Whitehead 2002). There was a dramatic decline in the number of females around the Galapagos Islands during 1985-1999 versus 1978-1992 levels, likely due to migration to nearshore waters of South and Central America (Whitehead 2003).

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

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

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

Sperm whales are sighted off Oregon in every season except winter (Green et al. 1992). However, sperm whales are found off California year-round (Barlow 1995; Dohl et al. 1983; Forney et al. 1995), with peak abundance from April to mid-June and from August to mid-

November (Rice 1974). Barlow (2003) reported mean group sizes of 2.0–11.8 during surveys the western U.S. Barlow (2003) estimated that 440 and 52 sperm whales occurred in Oregonian and Washingtonian waters, depending upon year and area, supported by densities of 0.0002 to 0.0019 individuals/km^2.

Hill and DeMaster (1999) concluded that about 258,000 sperm whales were harvested in the North Pacific between 1947-1987. Although the IWC protected sperm whales from commercial harvest in 1981, Japanese whalers continued to hunt sperm whales in the North Pacific until 1988 (Reeves and Whitehead 1997). In 2000, the Japanese Whaling Association announced plans to kill 10 sperm whales in the Pacific Ocean for research. Although consequences of these deaths are unclear, the paucity of population data, uncertainly regarding recovery from whaling, and re-establishment of active programs for whale harvesting pose risks for the recovery and survival of this species. Sperm whales are also hunted for subsistence purposes by whalers from Lamalera, Indonesia, where a traditional whaling industry has been reported to kill up to 56 sperm whales per year.

Southern Hemisphere. Whaling in the Southern Hemisphere averaged roughly 20,000 whales between 1956-1976 (Perry et al. 1999). Population size appears to be stable (Whitehead 2003). Whitehead (2002b) estimated 12,069 sperm whales south of 60° S.

Natural threats. Sperm whales are known to be occasionally predated upon by killer whales (Jefferson and Baird 1991; Pitman et al. 2001) and large sharks (Best et al. 1984) and harassed by pilot whales (Arnbom et al. 1987; Palacios and Mate 1996; Rice 1989b; Weller et al. 1996; Whitehead 1995). Strandings are also relatively common events, with one to dozens of individuals generally beaching themselves and dying during any single event. Although several hypotheses, such as navigation errors, illness, and anthropogenic stressors, have been proposed (Goold et al. 2002; Wright 2005), direct widespread causes remain unclear. Calcivirus and papillomavirus are known pathogens of this species (Lambertsen et al. 1987; Smith and Latham 1978).

Anthropogenic threats. Sperm whales historically faced severe depletion from commercial whaling operations. From 1800 to 1900, the IWC estimated that nearly 250,000 sperm whales were killed by whalers, with another 700,000 from 1910 to 1982 (IWC Statistics 1959-1983). However, other estimates have included 436,000 individuals killed between 1800-1987 (Carretta et al. 2005b). However, all of these estimates are likely underestimates due to illegal and inaccurate killings by Soviet whaling fleets between 1947 and 1973. In the Southern Hemisphere, these whalers killed an estimated 100,000 whales that they did not report to the IWC (Yablokov et al. 1998), with smaller harvests in the Northern Hemisphere, primarily the North Pacific, that extirpated sperm whales from large areas (Yablokov and Zemsky 2000). Additionally, Soviet whalers disproportionately killed adult females in any reproductive condition (pregnant or lactating) as well as immature sperm whales of either gender.

Following a moratorium on whaling by the IWC, significant whaling pressures on sperm whales were eliminated. However, sperm whales are known to have become entangled in commercial fishing gear and 17 individuals are known to have been struck by vessels (Jensen and Silber 2004). Whale-watching vessels are known to influence sperm whale behavior (Richter et al. 2006).

Sperm whales are also killed incidentally by gill nets at a rate of roughly nine per year (data from

1991 to 1995) in U.S. Pacific waters (Barlow et al. 1997). Sperm whales are known to interact (remove fish from) with longline fisheries in the Gulf of Alaska and entanglement has rarely been recorded (Hill and DeMaster 1999; Rice 1989a; Sigler et al. 2008). The stranding of two sperm whales along northern California revealed that 134 separate nets had been injected by the individuals, likely as free floating marine debris, leading to their deaths by gastric impaction (blockage) (Jacobsen et al. 2010).

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

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

Southern resident killer whale

Description of the species. Southern Resident killer whales compose a single population that occurs primarily along Washington State and British Columbia. The listed entity consists of three family groups, identified as J, K, and L pods.

Distribution. They are found throughout the coastal waters off Washington, Oregon, and Vancouver Island and are known to travel as far south as central California and as far north as the Queen Charlotte Islands, British Columbia. However, there is limited information on the range of Southern Residents along the outer Pacific Coast, with only 25 confirmed sightings of J, K, and L pods between 1982 and 2006 (Krahn et al. 2004a).

Movement and habitat. Southern Residents are highly mobile and can travel up to 100 miles per day (Baird 2000; Erickson 1978b). Members of K and L pods once traveled a straight-line distance of 584 miles from the northern Queen Charlotte Islands to Victoria, Vancouver Island, in seven days. Movements may be related to food availability.

Southern Resident killer whales spend a significant portion of the year in the inland waterways of the Strait of Georgia, Strait of Juan de Fuca, and Puget Sound, particularly during the spring, summer, and fall, when all three pods are regularly present in the Georgia Basin (defined as the Georgia Strait, San Juan Islands, and Strait of Juan de Fuca) (Felleman et al. 1991; Heimlich-Boran 1988; Olson 1998; Osborne 1999). Typically, K and L pods arrive in May or June and primarily occur in this core area until October or November. During this stay, both pods also make frequent trips lasting a few days to the outer coasts of Washington and southern Vancouver Island (Ford et al. 2000); however, J pod's movements differ considerably and are present only intermittently in the Georgia Basin and Puget Sound. Late spring and early fall movements of Southern Residents in the Georgia Basin have remained fairly consistent since the early 1970s,

with strong site fidelity shown to the region as a whole (NMFS 2005e). During late fall, winter, and early spring, the ranges and movements of the Southern Residents are less well known. Offshore movements and distribution are largely unknown for the Southern Resident population.

While the Southern Residents are in inland waters during the warmer months, all of the pods concentrate their activities in Haro Strait, Boundary Passage, the southern Gulf Islands, the eastern end of the Strait of Juan de Fuca, and several localities in the southern Georgia Strait (Felleman et al. 1991; Ford et al. 2000; Heimlich-Boran 1988; Olson 1998). Individual pods are similar in their preferred areas of use, although there are some seasonal and temporal differences in certain areas visited (Olson 1998). For example, J pod is the only group to venture regularly inside the San Juan Islands. The movements of Southern Resident killer whales relate to those of their preferred prey, salmon. Pods commonly seek out and forage in areas where salmon occur, especially those associated with migrating salmon (Heimlich-Boran 1986; Heimlich-Boran 1988; Nichol and Shackleton 1996).

Members of different pods do interact, but members generally remain within their matrilinear group (Parsons et al. 2009). However, additional interaction between pods has occurred over the past two decades, possibly in association with the decline of the Southern Resident population as a whole (Parsons et al. 2009).

Feeding. Southern Resident killer whales are fish eaters, and predominantly prey upon salmonids, particularly Chinook salmon but are also known to consume more than 20 other species of fish and squid (Ford and Ellis 2005; Ford and Ellis 2006; Ford et al. 2000; Ford et al. 1998; Saulitis et al. 2000; Scheffer and Slipp 1948). Killer whales show a strong preference for Fraser River Chinook salmon (78% of identified prey) during late spring to fall (Ford and Ellis 2006; Hanson et al. 2010b; Hanson et al. 2005). Chum salmon are also taken in significant amounts (11%), especially in autumn. Chinook are preferred despite much lower abundance in comparison to other salmonids (such as sockeye) presumably because of the species' large size, high fat and energy content, and year-round occurrence in the area. Killer whales also captured older (i.e., larger) than average Chinook (Ford and Ellis 2006). Throughout inland waters from May to September, Southern resident killer whale diet is approximately 88% Chinook (Hanson et al. 2007b; Hanson et al. 2010a), with a shift to chum salmon in fall. Little is known about the winter and early spring diet of Southern Residents. Early results from genetic analysis of fecal and prey samples indicate that Southern Residents consume Fraser River-origin Chinook, as well as salmon from Puget Sound, Washington and Oregon coasts, the Columbia River, and Central Valley of California (Hanson et al. 2007a; Hanson et al. 2010a). However, recent studies suggest that members of L pod have undergone dietary shifts from Chinook salmon during fall months over the past decade (Krahn et al. 2009a). Southern resident killer whales appear to be more sensitive to vessel disturbance while feeding than during other activities (Ashe et al. 2010). An area to the southwest of San Juan Island appears to be a foraging "hotspot" (Ashe et al. 2010)

Growth and reproduction. Female Southern Resident killer whales give birth to their first surviving calf between the ages of 12 and 16 years (mean ~ 14.9 years) and produce an average of 5.4 surviving calves during a reproductive life span lasting about 25 years (Matkin et al. 2003; Olesiuk et al. 1990a). Females reach a peak of reproduction around ages 20-22 and decline in calf production gradually over the next 25 years until reproductive senescence (Ward et al. 2009a). Older mothers tend to have greater calving success than do their younger, less-experienced counterparts (Ward et al. 2009b). Calving success also appears to be aided by the

assistance of grandmothers (Ward et al. 2009b). The mean interval between viable calves is four years (Bain 1990). Males become sexually mature at body lengths ranging from 17 to 21 feet, which corresponds to between the ages of 10 to 17.5 years (mean ~ 15 years), and are presumed to remain sexually active throughout their adult lives (Christensen 1984; Duffield and Miller 1988; Olesiuk et al. 1990a; Perrin and Reilly 1984a). Most mating is believed to occur from May to October (Matkin et al. 1997; Nishiwaki 1972; Olesiuk et al. 1990a). However, conception apparently occurs year-round because births of calves are reported in all months. Newborns measure seven to nine feet long and weigh about 440 lbs (Clark et al. 2000; Ford 2002; Nishiwaki and Handa 1958; Olesiuk et al. 1990a). Mothers and offspring maintain highly stable, life-long social bonds and this natal relationship is the basis for a matrilineal social structure (Baird 2000; Bigg et al. 1990; Ford et al. 2000). Some females may reach 90 years of age (Olesiuk et al. 1990a).

Diving. Killer whales tend to make relatively shallow dives. Of 87 tagged individuals in the Pacific Northwest, 31% of dives were less than 100 feet deep (Baird et al. 2003a). However, a free-ranging killer whale was recorded to dive to 264 m off British Columbia (Baird et al. 2005b). The longest duration of a recorded dive was 17 minutes (Dahlheim and Heyning 1999).

Status and trends. Southern Resident killer whales have been listed as endangered since 2005 (70 FR 69903). In general, there is little information available regarding the historical abundance of Southern Resident killer whales. Some evidence suggests that, until the mid- to late-1800s, the Southern Resident killer whale population may have numbered more than 200 animals (Krahn et al. 2002b). This estimate was based, in part, on a recent genetic study that found that the genetic diversity of the Southern Resident population resembles that of the Northern Residents (Barrett-Lennard 2000; Barrett-Lennard and Ellis 2001), and concluded that the two populations were possibly once similar in size. Unfortunately, lack of data prior to 1974 hinders long-term population analysis (NMFS 2005e). The only pre- 1974 account of Southern Resident abundance is from Sheffer and Slipp (1948) and merely notes that the species was "frequently seen" during the 1940s in the Strait of Juan de Fuca, northern Puget Sound, and off the coast of the Olympic Peninsula, with smaller numbers along Washington's outer coast. Olesiuk et al. (1990a) estimated the Southern Resident population size in 1967 to be 96 animals. Due to demand for marine mammals in zoos and marine parks, it is estimated that 47 killer whales, mostly immature, were taken from the Southern Resident population for public display between 1967 and 1973. By 1971, the level of removal decreased the population by about 30% to approximately 67 individuals (Olesiuk et al. 1990a). The population went then went through periods of decline and expansion for more than two decades. At the end of an 11-year growth cycle in 1995, the three Southern Resident pods – J, K, and L, reached a peak of 98 animals (NMFS 2008f).

More recently, the Southern Resident population has continued to fluctuate in numbers. After growing to 98 whales in 1995, the population declined by 17% to 81 whales in 2001 (-2.9% per year) before another slight increase to 84 whales in 2003 (Carretta et al. 2005a; Ford et al. 2000). The population grew to 90 whales in 2006, although it declined to 87 in 2007 (NMFS 2008f). The most recent population abundance estimate of 87 Southern Residents consists of 25 whales in J pod, 19 whales in K pod, and 43 whales in L pod (NMFS 2008f).

Natural threats. The recent decline, unstable population status, and population structure (e.g.,

few reproductive age males and non-calving adult females) continue to be causes for concern. Moreover, it is unclear whether the recent increasing trend will continue. The relatively low number of individuals in this population makes it difficult to resist/recover from natural spikes in mortality, including disease and fluctuations in prey availability (NMFS 2008f). Although disease outbreaks have not been identified in this population, increased contaminant load (see below) may increase the susceptibility of individuals to disease.

Anthropogenic threats. Numerous threats to the continued survival of Southern Resident killer whales have been identified (NMFS 2008f). Many of these are human in origin. The primary prey of killer whales, salmon, has been severely reduced due to habitat loss and overfishing of salmon along the West Coast (Gregory and Bisson 1997; Lackey 2003; Lichatowich 1999; NRC 1996; Pess et al. 2003; Schoonmaker et al. 2003; Slaney et al. 1996). Several salmon species are currently protected under the ESA, and are generally well below their former numbers. A 50% reduction in killer whale calving has been correlated with years of low Chinook salmon abundance (Ward et al. 2009a).

Puget Sound also serves as a major port and drainage for thousands of square miles of land. Contaminants entering Puget Sound and its surrounding waters accumulate in water, benthic sediments, and the organisms that live and eat here (Krahn et al. 2009a). As the top marine predator, Southern Resident killer whales bioaccumulate these toxins in their tissues, potentially leading to numerous physiological changes such as skeletal deformity, lowered disease resistance, and enzyme disruption (Baird 2001b; Calambokidis et al. 1984; Darnerud 2003; de Wit 2002; Hall et al. 2003; Hayteas and Duffield. 2000; Krahn et al. 2004b; Krahn et al. 2007; Krahn et al. 2009a; Krahn et al. 2004c; Krahn et al. 2002c; NMFS 2008e; Ross et al. 2000b; Ross et al. 2000c; Waring et al. 2004a; Ylitalo et al. 2001). Exposure is believed to be primarily through diet (Hickie et al. 2001); salmon preyed upon by SRKWs may also be at risk. Presently, the greatest contaminant threats are organochlorines, which include PCBs, pesticides, dioxins, furans, other industrial products, and the popularized chemical DDT (CBD 2001; Cullon et al. 2009; Krahn et al. 2009a; Krahn et al. 2002b; Ross et al. 2000a). These chemicals tend to bioaccumulate in fatty tissues, such as whale blubber, persist over long periods in the environment, and can be transmitted from mother to offspring (Haraguchi et al. 2009; Krahn et al. 2009a). Levels are much higher in field-sampled individuals than those found in a captive killer whale (Bennett et al. 2009). A similar, but separate concern is the growth of the petroleum industry in Puget Sound, which has the low potential to create a catastrophic oil spill, or more likely, small but chronic releases of petrochemicals. In addition, several hundred new chemicals enter the global marine environment annually; many of these chemicals have unknown effects to any lifeform (Grant and Ross. 2002; NMFS 2008e).

Encouragingly, many persistent organic pollutants have been banned in the U.S. and, over the past few decades, regulatory actions, Superfund clean-up, improved waste handling, and ongoing cleanup efforts have led to improvements in regional water quality(NMFS 2008e). This has led to decreasing levels of many organochlorine residues in the environment, (EVS Environmental Consultants 2003; Grant and Ross. 2002; Gray and Tuominen 2001; Mearns 2001) although it may take up to 60 years for some chemicals to fall below levels known to cause health effects in marine mammals (Hickie et al. 2001).

Vessel activity also has been identified as a threat. This includes physical harm or behavioral

modifications as well as habitat degradation/loss from U.S. naval vessel sonar activities, ship strike, and heavy and continuous presence by whale-watching vessels. In 2005, a U.S. vessel participating in sonar exercises apparently caused significant behavior changes in killer whale activity in the area, such that the whales vacated the area (NMFS 2005b). Although such activities are now receiving close scrutiny, the potential remains for these disruptions to occur, or as in other areas, the potential for auditory trauma, stranding, and death. The increase in "background noise" resulting from vessel traffic and coastal development activities, although not directly traumatic, has the potential to influence or disrupt the acoustic system that Southern Resident killer whales use to navigate, communicate, and forage (Bain and Dahlheim 1994; Erbe 2002c; Gordon and Moscrop 1996; Holt et al. 2009; NMFS 2008f; Williams et al. 2002a; Williams et al. 2002c). Commercial whale-watching in the region focuses primarily on Southern Resident killer whales and has increased dramatically in the recent years (Baird 2001a; Erbe 2002c; Koski 2004; Koski 2006b; Koski 2007a; MMMP 2002b; Osborne et al. 1999). Although mechanisms are in place to regulate the industry, concerns remain over persistent exposure to vessel noise, proximity to whales, which can cause behavioral changes, stress, or potentially the loss of habitat (Bain et al. 2006a; Bain et al. 2006c; Foote et al. 2004a; Kriete 2002; Kruse 1991; NMFS 2008f; Noren et al. 2009; Wiley et al. 2008; Williams et al. 2002a; Williams et al. 2002c).

Critical habitat. Critical habitat for the DPS of Southern Resident killer whales was designated on November 29, 2006 (71 FR 69054). Three specific areas were designated; (1) the Summer Core Area in Haro Strait and waters around the San Juan Islands; (2) Puget Sound; and (3) the Strait of Juan de Fuca, which comprise approximately 2,560 square miles of marine habitat. Three essential factors exist in these areas: water quality to support growth and development, prey species of sufficient quantity, quality, and availability to support individual growth, reproduction and development, as well as overall population growth, and passage conditions to allow for migration, resting, and foraging. Water quality has declined in recent years due to agricultural run-off, urban development resulting in additional treated water discharge, industrial development, oil spills. The primary prey of southern residents, salmon, has also declined due to overfishing and reproductive impairment associated with loss of spawning habitat. The constant presence of whale-watching vessels and growing anthropogenic noise background has raised concerns about the health of areas of growth and reproduction as well.

Cook Inlet beluga whale

Description of the species. Beluga whales are widely distributed in Arctic and subarctic waters, and in Alaska five putative populations exist (Beaufort Sea, eastern Chukchi Sea, Bristol Bay, eastern Bering Sea, and Cook Inlet)(Angliss et al. 2001). Cook Inlet beluga whales are the only population that is listed under the ESA. Mitochondrial and nuclear DNA distinguish Alaskan beluga whales from those that occur in Hudson Strait, Baffin Bay and the St. Lawrence River, with the Cook Inlet population demonstrating the strong evidence of genetic isolation from the other Alaskan populations and other populations demonstrating weak to moderate evidence of genetic isolation (O'Corry-Crowe 2008; O'Corry-Crowe et al. 2010; O'Corry-Crowe et al. 2007)**.**

Distribution. Beluga whales are observed year-round in Cook Inlet although less is known about their winter movements than summer movements (Hobbs et al. 2008). Data from satellite tagging studies suggest that movements of Cook Inlet beluga whales during summer months are short and largely focused around river estuaries and inlets (e.g., Chickaloon Bay, Turnagain Arm,

Susitna River, and Knik Arm in the upper inlet and in many cases the animals exhibited very little movement for weeks during the summer (Hobbs et al. 2005). Dense groupings in these areas during June and July are the focus of NMFS aerial surveys, but numbers drop substantially in the upper inlet by November (Hobbs et al. 2005). Outside of Cook Inlet in the Gulf of Alaska beluga whale sightings are extremely rare (Laidre et al. 2000). Hobbs et al. (2005) found that tagged beluga whales moved to farther offshore during winter months, but remained within Cook Inlet. However travel distance appeared to increase during winter months, and exhibited more widely dispersed patterns both within and among individuals (Hobbs et al. 2005). Distribution during all months is likely influenced by prey distribution, where salmon and eulachon are concentrated in river mouths during summer months and other prey like sand lance are found in mid and bottom waters of the inlet during winter months, albeit in more dispersed patterns leading to the wider dispersal of the whales.

Based on past studies of the summer distribution of beluga whales in Cook Inlet, it appears that the population has experienced a contraction in its overall distribution (Hobbs et al. 2008; Rugh et al. 2010; Speckman and Piatt. 2000). Aerial surveys in the 1970s indicated that at least 10% of the population used areas south of Kenai River and Kalgin Island (mid- to lower Cook Inlet) during summer months, whereas more recent surveys (1993-2007) observed more than 90% of the beluga whales in upper Cook Inlet in shallow waters. According to Hobbs et al. (2008) 90% of the whales in the 1970s were observed within 70 nmi of the western tip of Anchorage (Point Woronzof), whereas more recently (1998-2007) 90% were detected within 20 nmi. Although the precise reason for the range contraction is not known, the shrinking summer distribution likely reflects the reduction in the population size over the same intervals and the beluga whales' preference for dense aggregations of preferred prey species. Through the distribution has shrunk since the 1970s, possibly due to human hunting in excess of 50 individuals per year in the northern portions of Cook Inlet (where human disturbance is highest; city and port of Anchorage), surviving individuals seemed to re-occupy this habitat and abandon southern habitat (Rugh et al. 2010). This suggests that the northern reaches of Cook Inlet is superior habitat to southern areas, even with the anthropogenic stressors present there (Rugh et al. 2010). Overall, Cook Inlet beluga whale range has shrunk from greater than 7,000 km^2 to less than 3,000 km^2 (Rugh et al. 2010).

Habitat. During spring and summer months, beluga whales in Cook Inlet are typically concentrated near northern river mouths (Rugh et al. 2000). Cook Inlet experiences some of the most extreme tidal fluctuations in the world (see Hobbs et al. (2008) for a discussion), and beluga whales in the inlet have adapted to these tidal cycles and seemingly take advantage of them, although the precise causal reasons are not well known. Presumably, the feeding opportunities these tidal cycles proffer the beluga whale are a contributing factor. Beluga whales move further into inlets and arms during tidal floods and move back out during ebb tides in a relatively predictable manner, although highly-localized differences do exist (Ezer 2010). Tidal state is also believed to influence travel and milling behavior (Garner and Mckee. 2010). Aerial surveys and predictive models of habitat us indicate that beluga whale movement patterns are closely correlated to tidal patterns, flow accumulation and mudflats, with a preference for medium and high flow inlets of larger river basins (Ezer et al. 2008; Goetz et al. 2007). More information, however, is needed to link these habitat attributes to causative reasons for this preference. Besides feeding, studies have suggested this preference for tidal mudflats may also be attributed to calving and breeding, molting, or shelter from predators like killer whales (Calkins 1989a;

Huntington 2000b; Moore et al. 2000b; Shelden et al. 2003). Individual movements suggest utilization of a variety of areas, with little to no limitation to specific areas over time (Mcguire et al. 2010b).

Reproduction. Sexual maturity is believed to be attained at 4 to 10 years for females and at 8 to 15 for males (Nowak 1991; Suydam et al. 1999). Females typically produce a single calf every 2 to 3 years following a 14-month gestation. Of ten adult females examined after stranding, seven showed evidence of lactation, post-partum condition, or pregnancy when they died (Burek et al. 2010). Calving in Cook Inlet is assumed to occur from mid-May to mid-July, with some occurrences in April and August (Calkins 1983; Huntington 2000a; Mcguire et al. 2010a). Areas specific to calving and rearing have thus far not been identified, but movement to different areas within Cook Inlet has been noted; calves and their mothers appear to use the same habitats as other conspecifics (Mcguire et al. 2010a; Mcguire et al. 2010b). Alaska natives describe calving areas within Cook Inlet as the northern side of Kachemak Bay in April and May, off the mouths of the Beluga and Susitna Rivers in May, and in Chickaloon Bay and Turnagain Arm during the summer. According to surveys by LGL (Funk et al. 2005) cow/calf pairs also make extensive use of Knik Arm in the summer and fall. Neonates are often not seen until June in Cook Inlet (Burns and Seaman 1986a). The warmer waters from rivers may help keep newborns warm relative to cooler inlet waters (Calkins 1989b; Katona et al. 1983). Mating follows the calving period.

Calculation of beluga whale age is based on growth layers in teeth. Some debate exists as to whether a beluga whale tooth contains two growth layer groups (GLG) per year or one growth layer per year (Hobbs et al. 2008). Due to this ambiguity, Hobbs et al. (2008) summarized life history parameters according to tooth growth layers rather than years (Table 6 from (Hobbs et al. 2008)).

Table 6. Review of Female beluga life history parameters found in the published literature (Hobbs et al. 2008); GLG=growth layer groups).

Parameter	Data	Sources
Age at sexual maturity	8-15 GLG	(Braham 1984a; Brodie 1971; Burns and Seaman 1986b; Ognetov 1981; Seaman and Burns. 1981; Sergeant 1973)
	0% at 8-9 GLGs 33% at 10-11 GLGs 94% at 12-13 GLGs	(Burns and Seaman 1986b)[a]
	9.1 +/- 2.8 GLGs	(Robeck et al. 2005)
Age at 1st conception	54% at 8-9 GLGs 41% at 10-11 GLGs 94% at 12-13 GLGs	(Burns and Seaman 1986b)[b]
Age at senescence	42-43 GLGs	(Brodie 1971)
Pregnancy and birth rates	Small fetuses: 0.055 at 0-11 GLGs 0.414 at 12-21 GLGs 0.363 at 22-45 GLGs 0.267 at 46-57 GLGs 0.190 at 58-77 GLGs With full-term fetuses/neonates: 0.000 at 0-11 GLGs 0.326 at 12-21 GLGs 0.333 at 22-45 GLGs	(Burns and Seaman 1986b)

Parameter	Data	Sources
	0.278 at 46-51 GLGs	
	0.182 at 52-57 GLGs	
	0.125 at 58-77 GLGs	
Lifespan	>60 GLGs (Oldest female estimated at 70+ GLGs)	(Burns and Seaman 1986b)
	64-65 GLGs	(Khuzin 1961; Ohsumi 1979)
	60-61 GLGs	(Brodie 1971)
	50-51 GLGs	(Sergeant 1973)
Adult annual survival	0.96-0.97	(Beland et al. 1992)
	0.955 (based on pilot whale data)	(Brodie 1971)
	0.935	(Lesage and Kingsley. 1998)
	0.91-0.92	(Allen and Smith. 1978)
	0.906 (includes natural & human-caused mortality)	(Burns and Seaman 1986b)
	0.84-0.905 (based on body length and lifespan	(Ohsumi 1979)
Immature annual survival	0.905 (for neonates in first half year)	(Sergeant 1973)
Reproductive rate	0.010-012	(Perrin 1982)[c]
	0.11[d]	(Burns and Seaman 1986b)
	0.13[d]	(Sergeant 1973)
	0.09[d]	(Brodie 1971)
	0.09-0.12 [d]	(Braham 1984a)
	0.09-0.14[e]	(Braham 1984a)
	0.12 [e]	(Ray et al. 1984; Sergeant 1973)
	0.08-0.14[e]	(Davis and Evans 1982)
	0.06-0.10[e]	(Davis and Finley. 1979)
	0.08-0.10[e]	(Brodie et al. 1981)
	0.08 (unknown)	(Breton-Provencher 1980)
Calving Interval	<3 years	(Burns and Seaman 1986b)[f]
	2 yrs and 3 years	(Sergeant 1973)[g]

[a]Alaska sample (52 whales). Sampling occurring in June when most Alaskan beluga whales are born. Hobbs et al. (2008) note that it is possible that non-pregnant 8-9 GLGs beluga whales would have conceived before their 10-11 GLG birth date.
[b]Alaska sample of 22 whales.
[c]Based on literature review and adopted by the International Whaling Commission
[d]Based on annual calf production rates
[e]Based on calf counts
[f]For some female beluga whales. This was a tentative conclusion based on high conception rates noted in some females between the ages of 12-13 GLGs and 44-45 GLGs.
[g]Two-year intervals were for 25% of mature female belugas in eastern Canada (seven of 29 sampled); presumed after noting pregnancies occurred during lactation. Three-year intervals were for 75% of mature females in eastern Canada. Sergeant (1973) concluded that the "overlap of pregnancy and previous lactation is infrequent so that calving occurs about once in three years."

Lifespan. Lifespan is known to exceed 30 years of age (Burns and Seaman 1986a).

Feeding. Analyses of beluga whale stomach contents indicate that beluga whales are opportunistic feeders (including including octopi, squids, crabs, shrimps, clams, mussels, snails, sandworms, and fishes such as capelin, cod, herring, smelt, flounder, sole, sculpin, lamprey, lingcod, salmon, trout, whitefish, northern pike, grayling, and tomcod (Fay et al. 1984; Haley 1986; Huntington 2000a; Klinkhart 1966; Perez 1990)), but specific species form the bulk of the prey when they are seasonally abundant (Hobbs et al. 2008). For instance, eulachon (*Thaleichthys pacificus*) also known as smelt or candlefish, are a small anadromous fish return that their natal rivers in spring for spawning. In the Susitna River, the eulachon spawning migration has a bimodal peak, with fish entering the estuary in May and again in June, and represents a significant biomass of prey, with estimates of several thousand fish entering the river in the first wave and several million entering the river in June (Calkins 1989a). The common

name candlefish is derived from the fact the fish is so high in fat content during spawning, with up to 15% of total body weight as fat, that when caught and dried and strung on a wick the fish could be burned like a candle (Payne et al. 1999). This high fat content confers a significant source of energy for beluga whales, including calving whales that occur in the upper inlet during the same period (Calkins 1989a). The stomach contents of one beluga whale harvested in upper Cook Inlet in 1998 near the Susitna River contained only eulachon. Based on stomach sample analyses from 2002-2007 fish compose the majority of the prey species, with gadids (cod and walleye pollock) and salmonids composing the majority of the fish eaten (Hobbs et al. 2008). Anadromous salmonids begin concentrating at the river mouths and intertidal flats in upper Cook Inlet in late spring and early summer as emigrating smolts and immigrating adult spawners (Fried et al. 1979; Hazard 1988). Like eulachon, salmon are another source of lipid-rich prey for the beluga whale and represent the greatest percent frequency of occurrence of the prey species found in Cook Inlet beluga whale stomachs(Hobbs et al. 2008). As salmonid numbers dwindle in the fall and winter, beluga whales return to feed on nearshore or deeper water species including cod, sculpin, flounder, sole, shrimp, crab and others (Hobbs et al. 2008). Feeding frequently occurs collaboratively. Significantly, it appears that Cook Inlet beluga whales have switched diets during 1965 to 2007(Quakenbush and Nelson. 2010). Although the exact nature of this switch is unknown, it appears that individual whales are currently feeding at a lower trophic level than they formerly did(Quakenbush and Nelson. 2010).

Status and trends. On October 22, 2008, NMFS listed the Cook Inlet beluga whale as endangered (73 FR 62919). The Cook Inlet population is relatively isolated compared to other beluga whale stocks and the Alaska Peninsula is likely an effective physical barrier to genetic exchange, perhaps for thousands of years (Laidre et al. 2000; Murray and Fay 1979; O'Corry-Crowe et al. 1997). As such, this population may be particularly sensitive to decline. Because few robust data are available prior to the 1990s, historical population size is based upon surveys lacking strong statistical confidence (Table 7). These estimates range from 150 to 450 individuals between 1963 and 1988 (Calkins 1984; Hazard 1988; Klinkhart 1966; Murray and Fay 1979; Sergeant and Brodie 1975). However, the most robust survey prior to 1994 occurred in 1979 and estimated Cook Inlet to contain 1,293 individuals and may have been an underestimate of the population (Calkins 1989b). Based upon this survey, the maximum stable size of the Cook Inlet population is believed to be 1,300 individuals and this value is used for management purposes.

Table 7. Estimated abundance of Cook Inlet beluga whales with coefficient of variation and 95% confidence intervals.

Year	Estimate[1]	CV	95% CI[2]	
			Lower	Upper
1979	1,293			
1994	653	0.43	291	1464
1995	491	0.44	215	1120
1996	594	0.28	347	1018
1997	440	0.14	335	578
1998	347	0.29	199	606
1999	367	0.14	279	482
2000	435	0.23	279	679
2001	386	0.087	326	458

2002	313	0.12	248	396
2003	357	0.107	290	440
2004	366	0.2	290	440
2005	278	0.18	196	394
2006	302	0.16	221	412
2007	375	0.14	285	492
2008	375	0.23	240	585
2009[2]	321	0.18	226	456

[1]All estimates, except 1979 estimate, reported in Hobbs & Shelden (2008). The 1979 estimate is from Calkins (1989a) as cited in NMFS 2008.
[2]Data from R. Hobbs, pers. comm., to A. Garrett, Apr. 2010.

Systematic aerial surveys of beluga whales throughout Cook Inlet began in 1993. Surveys each June from 1999 to 2005 have estimated abundance between 278 and 435 individuals, with a sharp 47% decline between 1994 and 1998 proceeding that (Hobbs et al. 2000, NMFS unpublished data). Data from June 2006 surveys supported an abundance of 302 individuals (Angliss and Outlaw 2007a). Trends support a 71% probability that the population is declining and this trend is likely to continue (Hobbs et al. 2006; Lowry et al. 2006).

Between 1979 and 1994, according to above noted population estimates, Cook Inlet beluga whales declined by 50%, with another 50% decline observed between 1994 and 1998. Using a growth fitted model Hobbs et al. (2008) observed an average annual rate of decline of -2.91% (SE = 0.010) from 1994 to 2008, and a -15.1% (SE=0.047) between 1994 and 1998. A comparison with the 1999-2008 data suggests the rate of decline at -1.45% (SE=0.014) per year (Hobbs et al. 2008). Reasons for population decline are somewhat nebulous, but mortality pressures include native harvests, strandings, and killer whale predation (Angliss and Outlaw 2007a; Mahoney and Shelden 2000). However, Cook Inlet harvests have been severely restricted (1-2 whales annually) since 1999, which coincides with a significant trend in the stabilization of the beluga whale population. Despite this, the population has not yet shown signs of recovery to historical numbers. Although not currently factors, disease and habitat disturbance have the potential to significantly impact the Cook Inlet population. According to the best available population viability analysis, there is an 80% probability that the population is declining, a 26% probability that the population will be extinct in 100 years (by 2108) and a 70% probability that the population will be extinct within 300 years (by 2308).

Natural threats. Cook Inlet beluga whales have significant natural mortality from a variety of sources (NMFS 2006c). According to Hobbs et al. (2008), over 700 beluga whales have stranded in Cook Inlet since 1988. Although many of these strandlings survive and refloat with the next tide, a portion do not (Mahoney et al. 2010). Killer whales are known to prey upon beluga whales in Cook Inlet and although exact removal numbers are unknown, small reductions in a small population can limit recovery (Burek et al. 2010; Mahoney et al. 2010). Herpes virus, meningoencephalitis, and parasites have been identified as diseases in Cook Inlet beluga whales (Burek et al. 2010). However, the significance of disease and other causes of death is difficult to estimate due to the poor condition in which most dead-stranded beluga whales are found in Cook Inlet (Burek et al. 2010). Loss of genetic variability has been raised as a possible reason for the failure of Cook Inlet beluga whales to recover; studies to evaluate this possibility are currently underway (Bechdel et al. 2010).

Anthropogenic threats. Cook Inlet beluga whales have been exposed to anthropogenic pressures in the past, including incidental takes in gillnet fisheries of roughly three to six individuals per year (Burns and Seaman 1986a; Murray and Fay 1979). During the early 1900s there was a short-lived commercial whaling company, The Beluga Whaling Company, which operated at the Beluga River in upper Cook Inlet. The Company during its 5 years of operation harvest 151 belugas from 1917-1921 (Mahoney and Shelden. 2000). Although evidence does not presently exist for harassment of belugas, areas of high beluga use coincide with regions of lowest ambient sound and have been known to evacuate high-use areas of humans (Huntington 2000a). Significant behavioral changes have been observed in Cook Inlet belugas in response to small boat proximity (NMFS 2006c). Propeller scars are observed on belugas, although direct mortality has not been verified.

Although belugas in the St. Lawrence Seaway and Canadian Arctic have high contaminant burdens and subsequent biological impairments (such as cancer), belugas in Cook Inlet have been found to have low concentrations of PCBs, organochlorines, and heavy metals (Becker et al. 2000a; Kelly et al. 2009a; Mcaloose and Newton. 2009; Tomy et al. 2009), although in situ concentrations of some organic pollutants have increased over the past 20 years (Becker 2010). However, copper is two to three times higher in Cook Inlet beluga whales than beluga whales from the eastern Beaufort Sea and the eastern Chukchi Sea, but is similar concentrations found in Hudson Bay beluga whales (Becker et al. 2000b). Studies indicate that PCBs and chlorinated pesticide concentrations are higher in male St. Lawrence Seaway beluga whales than females, reflecting the transference of body loads to the offspring that occurs during gestation and lactation (Becker et al. 2000b). However, discharges from industrial activities that do not enter municipal treatment systems (petroleum, seafood processing, ship ballast, dredging), discharges from municipal wastewater treatment systems, runoff from urban areas, mining operations, military sites, airports and agricultural areas, and accidental spills or discharges of petroleum and other products remain concerns for stock recovery (Moore et al. 2000a; NMFS 2006c). Selenium from tooth samples has been found to be a good predictor of selenium levels in liver and muscle (selenium can be a predictor of mercury levels) (Kinghorn et al. 2008). Mercury levels also vary by age, as individuals grow older and switch dietary preferences (Loseto and Ferguson 2008).

Critical habitat. NMFS proposed critical habitat for the Cook Inlet beluga whale on December 2, 2009 (74FR 63080). Two areas specific areas are proposed comprising 7,809 square kilometers of marine habitat. Area 1 encompasses 1,918 square kilometers (741 sq. mi.) of Cook Inlet northeast of a line from the mouth of Threemile Creek (61º 08.5'N., 151 º 04.4' W.) to Point Possession (61º 02.1'N., 150 º 24.3' W.). This area is bounded by Anchorage, the Matansuska-Susitna Borough, and the Kenai Peninsula Borough. This area contains shallow tidal flats, river mouths, or estuarine areas and is important as foraging and calving habitats. Area 1 also has the highest concentrations of beluga whales in the spring through fall as well as the greatest potential for adverse impact from anthropogenic threats. Area 1 contains many rivers with large eulachon and salmon runs, including two rivers in Turnagain Arm (Twenty-mile River and Placer River) which are visited by beluga whales in the early spring. Use declines in the summer and increases again in August through the fall, coinciding with Coho salmon returns. Also included in Area 1 are Knik Arm and the Susitna delta. Area 2 consists of 5,891 square kilometers (2,275 sq. mi.) of Cook Inlet, located south of Area 1, north of a line at 60º 25.0'N., and includes nearshore areas south of 60º 25.0'N along the west side of the Inlet and Kachemak Bay on the east side of the lower inlet. Area 2 is used by Cook Inlet beluga whales in a dispersed

fashion for fall and winter-feeding and as transit waters. Area 2 includes near and offshore areas of the mid and upper Inlet, and nearshore areas of the lower Inlet. Area 2 includes Tuxedni, Chinitna, and Kamishak Bays on the west coast and a portion of Kachemak Bay of the east coast. Dive studies indicate that beluga whales in this area dive to deeper depths and are at the surface less frequently than they are when they inhabit Area 1. The primary constituent elements essential to the conservation of Cook Inlet beluga whales are: (1) intertidal and subtidal waters of Cook Inlet with depths <30 ft. (MLLW) and within 5 miles of high and medium flow accumulation anadromous fish streams; (2) primary prey species consisting of four species of Pacific salmon (Chinook, Coho, sockeye, and chum salmon), Pacific eulachon, Pacific cod, walleye pollock, saffron cod, and yellowfin sole; (3) the absence of toxins or other agents of a type or amount harmful to beluga whales; (4) Unrestricted passage within or between the critical habitat areas; and (5) absence of in-water noise at levels result in the abandonment of habitat by Cook Inlet beluga whales. The comment period on this proposed rule closed on February 1, 2010.

Pinnipeds

Steller sea lion

Description of the species. Steller sea lions are distributed along the rim of the North Pacific Ocean from San Miguel Island (Channel Islands) off Southern California to northern Hokkaido, Japan (Loughlin et al. 1984; Nowak 2003). Their centers of abundance and distribution are in Gulf of Alaska and the Aleutian Islands, respectively (NMFS 1992). In the Bering Sea, the northernmost major rookery is on Walrus Island in the Pribilof Island group. The northernmost major haul-out 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. For management purposes, two stocks have been designated, but which represent a single population. These stocks likely have some taxonomic basis at the sub-species level in both genetics and skull morphology (Phillips et al. 2009).

Distribution. The eastern DPS of Steller sea lions includes animals east of Cape Suckling, Alaska (144°W) south to California waters (55 FR 49204). The western DPS of Steller sea lion includes animals west of Cape Suckling, Alaska (144°W; 62 FR 24345). Most adult Steller sea lions occupy rookeries during the pupping and breeding season and exhibit a high level of site fidelity. During the breeding season, some juveniles and non-breeding adults occur at or near the rookeries, but most are on haulouts (sites that provide regular retreat from the water on exposed rocky shoreline, gravel beaches, and wave-cut platforms or ice; (Ban 2005; Call and Loughlin 2005; Rice 1998a). Adult males may disperse widely after the breeding season. Males that breed in California move north after the breeding season and are rarely seen in California or Oregon except from May through August (Mate 1973). During fall and winter many sea lions disperse from rookeries and increase use of haulouts, particularly on terrestrial sites but also on sea ice in the Bering Sea. The rookeries off southern Oregon are located along the coast at Rogue and Orford reefs near 42º25’ and 42º45’N and 124º30’W, respectively (Bonnell et al. 1992). Counts of adults and juveniles in Oregon have shown a gradual increase from 1486 in 1976 to 3648 in 2001 (Angliss and Outlaw 2005b).

Reproduction. Female Steller sea lions reach sexual maturity and first breed between three and eight years of age and the average age of reproducing females (generation time) is about 10 years

(Calkins and Pitcher 1982; Pitcher and Calkins 1981; York 1994). They give birth to a single pup from May through July and then breed about 11 days after giving birth. Females normally ovulate and breed annually after maturity although there is a high rate of reproductive failures. The gestation period is believed to be about 50 to 51 weeks (Pitcher and Calkins 1981). The available literature indicates an overall reproductive (birth) rate on the order of 55% to 70% or greater (Gentry 1970; Pike and Maxwell 1958; Pitcher and Calkins 1981). However, natality was reported to be low in the western DPS in recent years (2003-2009; 69%) versus earlier years (43%); (Maniscalco et al. 2010). Survival through the first three weeks can be less than 50% at some sites, while others can be over 90% (Kaplan et al. 2008). Twinning has been reported (Maniscalco and Parker. 2009).

Males reach sexual maturity at about the same time as females (three to seven years of age, reported in (Loughlin et al. 1987), but generally do not reach physical maturity and participate in breeding until about eight to ten years of age (Pitcher and Calkins 1981). The sex ratio of pups at birth is assumed to be about 1:1 or biased toward slightly greater production of males, but non-pups are biased towards females (Calkins and Pitcher 1982; NMFS 1992; Pike and Maxwell 1958; Trites and Larkin 1992; York 1994).

Mothers with newborn pups will make their first foraging trip about a week after giving birth, but trips are short in duration and distance at first, then increase as the pup gets older (Maniscalco et al. 2006; Merrick and Loughlin 1997; Milette 1999; Milette and Trites 2003; Pitcher et al. 2001). Females attending pups tend to stay within 20 nm of the rookery (Calkins 1996; Merrick and Loughlin 1997). Newborn pups are wholly dependent upon their mother for milk during at least their first three months of life, and observations suggest they continue to be highly dependent upon their mother through their first winter (Porter 1997; Scheffer 1945; Trites et al. 2006). Generally, female Steller sea lion will nurse their offspring until they are one to two years old (Calkins and Pitcher 1982; Gentry 1970; Pitcher and Calkins 1981; Sandegren 1970; Trites et al. 2006).

Habitat. Steller sea lions are not known to make regular migrations but do move considerable distances. Adult males may disperse hundreds of miles after the breeding season (Calkins 1986; Calkins and Pitcher 1982; Loughlin 1997). Adult females may travel far out to sea into water greater than 3,300 feet deep (Merrick and Loughlin 1997). Studies on immature Steller sea lions indicate three types of movements: long-range trips (greater than 9.3 miles and greater than 20 hours), short-range trips (less than 9.3 miles and less than 20 hours), and transits to other sites (NMFS 2007b). Long-range trips started around 9 months of age and likely occur most frequently around the time of weaning, while short-range trips happen almost daily. Young individuals generally remain within 300 miles of rookeries their first year before moving further away in subsequent years (Raum-Suryan et al. 2004). Many animals also use traditional rafting sites, which are places where they rest on the ocean surface in a tightly packed group (Bigg 1985)NMFS unpublished data). Frontal features with small-scale temperature gradients appear to be attractive foraging sites for juvenile Steller sea lions (Lander et al. 2010).

Feeding. Steller sea lions are generalist predators that eat various fish (arrowtooth flounder, rockfish, hake, flatfish, Pacific salmon, Pacific herring, Pacific cod, sand lance, skates, cusk eel, lamprey, walleye, Atka mackerel), squids, and octopus and occasionally birds and marine mammals (Brown et al. 2002; Calkins and Goodwin 1988; Daniel and Schneeweis 1992; Jones

1981; McKenzie and Wynne 2008; Olesiuk et al. 1990b; Pitcher and Fay 1982; Sinclair and Zeppelin 2002; Womble and Conlon. 2010). Diet is likely strongly influenced by local and temporal changes in prey distribution and abundance (McKenzie and Wynne 2008; Sigler et al. 2009). Haulout selection appears to be driven at least in part by local prey density (Winter et al. 2009).

Diving. Diving activity is highly variable in Steller sea lion by sex and season. During the breeding season, when both males and females occupy rookeries, adult breeding males rarely, if ever, leave the beach (Loughlin 2002). However, females tend to feed at night on one to two day trips and return to nurse pups (NRC 2003a). Female foraging trips during winter are longer (80 miles) and dives are deeper (frequently greater than 820 feet). Summer foraging dives, however, are closer to shore (about 10 miles) and shallower (330 to 820 feet; (Loughlin 2002; Merrick and Loughlin 1997). As pups mature and start foraging for themselves, they develop greater diving ability until roughly 10 years of age (Pitcher et al. 2005). Juveniles usually make shallow dives to just over 50 feet, but much deeper dives in excess of 1,000 feet are known (Loughlin et al. 2003). Young animals also tend to stay in shallower water less than 330 feet deep and within a dozen miles from shore (Fadely et al. 2005).

Status and trends. Steller sea lions were originally listed as threatened under the ESA on November 26, 1990 (55 FR 49204), following a decline in the U.S. of about 64% over previous three decades. In 1997, the species was split into two separate populations based on demographic and genetic differences (Bickham et al. 1996; Loughlin 1997), and the western population was reclassified to endangered (62 FR 24345) while the eastern population remained threatened (62 FR 30772). The Steller sea lion is also listed as endangered on the 2007 IUCN Red List (Group 1996).

Loughlin et al.(1984) estimated the worldwide population of Steller sea lions was between 245,000 and 290,000 animals (including pups) in the late 1970s. Though the genetic differences between the eastern and western DPSs were not known at the time, Loughlin et al. (1984) noted that 90% of the worldwide population of Steller sea lions was in the western DPS in the early 1980s (75% in the U.S. and 15% in Russia) and 10% in the eastern DPS. Loughlin et al. (1984) concluded that the total worldwide population size (both DPSs) was not significantly different from that estimated by Kenyon and Rice (1961) for the years 1959 and 1960, though the distribution of animals had changed. Steller sea lions collected in the Gulf of Alaska during the early 1980s showed evidence of reproductive failure and reduced rates of body growth that were consistent with nutritional limitation (Calkins et al. 1998; Calkins and Goodwin 1988; Pitcher et al. 1998). After conducting a range-wide survey in 1989, Loughlin et al. (1992) noted that the worldwide Steller sea lion population had declined by over 50% in the 1980s, to approximately 116,000 animals, with the entire decline occurring in the range of the western DPS.

Western DPS-The western stock appears to be in decline. Between late 1970s and the mid-1990s, counts of the western population of sea lions fell from 109,880 animals to 22,167 animals, a decline of 80% (Hauser et al. 2007; NMFS 1995). The 1996 count was 27% lower than the count in 1990. Fritz and Stinchcomb (2005) estimate that from 1991 to 2000, the number of adults and juvenile sea lions in the western population declined by about 38%. Surveys by Fritz and Stinchcomb (2005) indicate that the current number of non-pups in the western population is 29,037. NMFS combined this number with the number of pups in 2004-2005 (9,951) to reach

the current minimum population estimate of 38,988 of Steller sea lion in the western U.S. (the western stock); when combined with data on Steller sea lions in Russia the minimum population estimate is 44,780 (Angliss and Outlaw 2007a).

A number of population models have been developed for Steller sea lions (Gerber and VanBlaricom 2001; Goodman 2006; Holmes and York 2003; Pascual and Adkison 1994; Winship and Trites 2006; York et al. 1996). According to several population models the western DPS has significant chance of going extinct within the next 100 years (Goodman 2006; Winship and Trites 2006; York et al. 1996), while many individual rookeries (breeding aggregations) however, have a much higher risk of extinction (e.g., western Aleutian island rookeries and Gulf of Alaska) (Winship and Trites 2006).

Eastern DPS-The eastern stock seems to be more optomistic than the western stock. Trend counts in Oregon were relatively stable in the 1980s, showing a gradual increase in numbers since 1976 (NMFS 2005d). Numbers in California, however, have declined to less than 2,000 non-pups, from counts between 1927 and 1947 that were as high as 7,000 non-pups (NMFS 2005d). The count from Central California in 2000, reached the second lowest count of 349 non-pups (in 1992 the count was as low as 276 non-pups). In Southeast Alaska, counts of non-pups at trend sites increased by 56% from 1979 to 2002 from 6,376 animals to 9,951 (NMFS 2005d; Sease et al. 2001). Counts of non-pups at British Columbia trend sites increased nearly 260% between 1982 and 2002 (NMFS 2005d).

NMFS considers this population to be increasing, and multiplies pup counts by a factor of 4.5 (based on (Calkins and Pitcher 1982) or 5.1 (Trites and Larkin 1996) to estimate the total population size (Angliss and Outlaw 2008). Pup count data from 2002 through 2005 from across the range of the eastern population, multiplied by a factor of 4.5 or 5.1 results in a population estimate of 48,519 or 54,989 animals. In 2005, 5,510 pups were counted in Alaska, 3,318 pups were counted in British Columbia in 2002, 1,136 pups were counted in Oregon in 2002, and 818 counted in California in 2004. The current minimum population estimate is 44,584 animals. NMFS calculates this estimate by adding non-pup counts taken in 2002 in Southeast Alaska, to counts of animals in Washington in 2002 as well as counts of pups and non-pups in Canada in 1998, Oregon in 2002, California in 2004, and southeastern Alaska in 2005 (Angliss and Outlaw 2008).

Off Oregon and Washington, nearly 90% of Steller sea lion sightings have occurred within 21 km of shore and none further out than 40 km or in waters greater than 200 m deep (Bonnell et al. 1992). In the fall (September and November surveys), mean density was 0.011/km^2 (Bonnell et al. 1992).

Estimated annual mortality is 0.22 for ages 0-2, dropping to 0.07 at age 3, then increasing gradually to 0.15 by age 10 and 0.20 by age 20 (York 1994). Population modeling suggested that decreased juvenile survival likely played a major role in the decline of sea lions in the central Gulf of Alaska during 1975-1985 (Holmes and York 2003; Pascual and Adkison 1994; York 1994).

Natural threats. Killer whale predation, particularly on the western DPS under reduced population size, may cause significant reductions in the stock (NMFS 2008g). Sleeper sharks are also significant predators of Steller sea lions. Frid et al. (2009) suggested that risk of predation in

nearshore waters by killer whales and offshore predation risk by sleeper sharks limited the use of Pacific herring in deep water and walleye Pollock in shallow water.

Steller sea lions have tested positive for several pathogens, but disease levels are unknown (FOC 2008). Similarly, parasites in this species are common, but mortality resulting from infestation is unknown. However, significant negative effects of these factors may occur in combination with stress, which reduces immune capability to resist infections and infestations. If other factors, such as disturbance, injury, or difficulty feeding occur, it is more likely that disease and parasitism can play a greater role in population reduction.

Anthropogenic threats. Steller sea lions were historically and recently subjected to substantial mortality by humans, primarily due to commercial exploitation and both sanctioned and unsanctioned predator control, (Atkinson et al. 2008; Bigg 1988; Bonnot 1928; Bonnot and Ripley 1948; NMFS 2008g; Pearson and Verts 1970; Rowley 1929; Scheffer 1945; Scheffer 1950). Several dozen individuals may become entangled and drown in commercial fishing gear (Atkinson et al. 2008; NMFS 2008g). Several hundred individuals are removed by subsistence hunters annually in controlled and authorized harvests. Occasional harvest occur in Canada (FOC 2008). Additional mortality (362 from 1990 to 2003) has occurred from shooting of sea lions interfering in aquaculture operations along British Columbia (FOC 2008). Marine debris is also concerning for the health of Steller sea lion populations. It is estimated that 0.26% of Steller sea lions have marine debris around their necks or are hooked by fishing gear (0.07%) (FOC 2008; Raum-Suryan et al. 2009).

Significant concern also exists regarding competition between commercial fisheries and Steller sea lions for the same resource: stocks of pollock, Pacific cod, and Atka mackerel. Significant evidence exists that supports the western DPS declining as a result of change in diet and resulting declines in growth, birth rates, and survival (Atkinson et al. 2008; Calkins et al. 1998; Calkins and Goodwin 1988; Pitcher et al. 1998; Trites and Donnelly 2003). As a result, limitations on fishing grounds, duration of fishing season, and monitoring have been established to prevent Steller sea lion nutritional deficiencies as a result of inadequate prey availability.

Behavioral disruption occurs as a result of human disturbance (FOC 2008). Research efforts to collect scats, count and weigh pups, and other human activities on or near rookeries can lead to stampedes into the water. Mortality can occur directly due to pup trampling, separating from mothers, or drowning. If disturbance is too frequent, haulouts may be completely abandoned. Although habituation to some activities, such as boating, can occur, unusual activities and sounds, such as blasting or demolition, can remotely trigger stampedes.

Contaminants are a considerable issue for Steller sea lions. Roughly 30 individuals died as a result of the Exxon *Valdez* oil spill and contained particularly high levels of PAH contaminants, presumable as a result of the spill. Blood testing confirmed hydrocarbon exposure. Subsequently, premature birth rates increased and pup survival decreased (Calkins et al. 1994; Loughlin et al. 1996). Organochlorines, including PCBs and DDT (including its metabolites), have been identified in Steller sea lions in greater concentrations than any other pinniped during the 1980s, although levels appear to be declining (Barron et al. 2003; Hoshino et al. 2006). The levels of PCBs have been found to have twice the burden in individuals from Russia than from western Alaska (4.3 ng/g wet weight versus 2.1 ng/g wet weight; (Myers et al. 2008). Levels of DDT in Russian pups were also on average twice that in western Alaska pups (3.3 ng/g wet

weight blood versus 1.6 ng/g wet weight). The source of contamination is likely from pollack, which have been found to contain organochlorines throughout the Gulf of Alaska, but higher in regions occupied by the eastern DPS of Steller sea lions (NMFS 2008g). Heavy metals, including mercury, zinc, copper, metallothionien, and butyltin have been identified in Steller sea lion tissues, but are in concentrations lower than other pinnipeds (Beckmen et al. 2002; Castellini 1999; Kim et al. 1996; NMFS 2008g; Noda et al. 1995). Mercury may be of higher significance, with liver levels being measured at levels above those necessary to impact fish (Holmes et al. 2008). However, contaminants leading to mortality in Steller sea lions have not been identified (NMFS 2008g). Contaminant burdens are lower in females than males, because contaminants are transferred to the fetus *in utero* as well as through lactation (Lee et al. 1996; Myers et al. 2008). However, this means that new generations tend to start with higher levels of contaminants than their parents originally had. Concerns over Steller sea lion contaminants are of additional concerns because contaminants in the body tend to be mobilized as fat reserves are used, such as when prey availability is low; a situation that is likely occurring for Steller sea lions today.

Critical habitat. Critical habitat was designated on August 27, 1993 for both eastern and western DPS Steller sea lions in California, Oregon, and Alaska (58 FR 45269). Steller sea lion critical habitat includes all major rookeries in California, Oregon, and Alaska and major haulouts in Alaska. Essential features of Steller sea lion critical habitat include the physical and biological habitat features that support reproduction, foraging, rest, and refuge, and include terrestrial, air and aquatic areas. Specific terrestrial areas include rookeries and haul-outs where breading, pupping, refuge and resting occurs. More than 100 major haulouts are documented. The principal, essential aquatic areas are the nearshore waters around rookeries and haulouts, their forage resources and habitats, and traditional rafting sites. Air zones around terrestrial and aquatic habitats are also designated as critical habitat to reduce disturbance in these essential areas. Specific activities that occur within the habitat that may disrupt the essential life functions that occur there include: (1) wildlife viewing, (2) boat and airplane traffic, (3) research activities, (4) timber harvest, (5) hard mineral extraction, (6) oil and gas exploration, (7) coastal development and pollutant discharge, and others.

In addition, British Columbia has established protective areas in which Steller sea lion rookeries occur at Triangle Island and Cape St. James (Canada 2008). Several other haul-out sites occur within Canadian national and provincial parks. Further, the Canadian government is moving to establish a marine wildlife area for the Scott Islands, where Steller sea lions haul-out and breed.

Proposed species

False killer whale-Hawaiian insular DPS

Description of the species. Hawaiian insular false killer whales (HIFKWs) are genetically unique compared to the pelagic form in surrounding Pacific waters; at a broader level, individuals inhabiting the Central Pacific are genetically different from those in the Eastern Pacific (Chivers et al. 2010; Chivers et al. 2007). Genetic data suggest little immigration into the HIFKW population. Additional data are being collected to identify whether other false killer whale groups are part of the Hawaiian insular population.

Distribution. HIFKWs move widely and rapidly among the main Hawaiian Islands, traveling up

to 112 km from shore over a total range of 77,600 km^2 (Baird 2009; Baird et al. 2008; Baird et al. 2005a; Baird et al. 2010; Forney et al. 2010; Oleson et al. 2010). Individuals can move between islands within a matter of days (Oleson et al. 2010). However, they do not appear to move broadly within the ocean basin, as is generally assumed for false killer whales. Part of HIFKW range overlaps with pelagic forms of false killer whales between 42 and 112 km from shore (Baird et al. 2010; Forney et al. 2010).

Growth and reproduction. False killer whales generally reach sexual maturity at 8-11 years of age for females and 8-10 years for males (Kasuya 1986; Odell and McClune. 1999; Stacey et al. 1994). Individuals grow to 40-50% of adult body length in their first year, but males continue to grow faster and to a larger size thereafter (Kasuya 1986). This leads to a degree of sexual dimorphism, with males larger in size than females, the degree of which varies around the world; in Japan, females are about 84% the length of males (Ferreira 2008; Kitchener et al. 1990). Maximum body size appears to vary at different locations, although growth appears to end after 20-30 years of age (Ferreira 2008; Kasuya 1986). Data from Japanese drive fisheries found a nearly 2:1 sex bias towards females (Ferreira 2008).

There is debate regarding false killer whale mating systems, which may be polygamous or matrilinear (Ferreira 2008). Females ovulate at least annually, apparently at random, and calving can occur year-round (Stacey et al. 1994). Ovulation rates decrease with age to the point that females over the age of 44 years are considered reproductively senescent (Ferreira 2008; Kasuya 1986) rates for false killer whales have been estimated at 14-21% of females annually, although this has been found to vary (11.4% in Japan and 2.2% in South Africa)(Kasuya 1986; Perrin and Reilly 1984b). Gestation lasts 11-16 months in captivity (Brown et al. 1966). Lactation lasts 18-24 months (Perrin and Reilly 1984b). Calving intervals have been estimated at roughly 7-9 years in Japan (Ferreira 2008; Stacey et al. 1994), relatively long for cetaceans. However, this varies, with 4.5 years in South Africa (Ferreira 2008).

Maximum lifespan for false killer whales has been reported as 63 years for females and 58 for males (Kasuya 1986). Some individuals have been resighted in Hawaiian waters over a 21-year time span (Baird et al. 2008).

Behavior. False killer whale group sizes can vary widely. Group sizes average 10-30 individuals based upon aerial and vessel surveys, but groups stranding on shore are generally much larger, frequently numbering from 100 to more than 800 individuals (Baird 2009; Baird et al. 2008; Baird et al. 2010; Ferreira 2008; Ross 1984; Wade and Gerrodette 1993). It has been proposed that groups seen during surveys are a part of larger aggregations maintaining acoustic contact (Baird et al. 2010). Indeed, larger dispersed aggregations of false killer whales have been noted during surveys (Baird 2009; Carretta et al. 2007b; Reeves et al. 2009b; Wade and Gerrodette 1993) that can move in a coordinated fashion (Baird et al. 2008). HIFKWs form strong long-term bonds (Baird et al. 2008)

Diving is not well-known in false killer whales, but individuals are believed capable of reaching 500 m in depth and possibly 700 m (Cummings and Fish. 1971; Oleson et al. 2010). However, most dives are significantly shallower. HIFKWs occasionally dive to 150 m (apart from the possible 700 m dive), with frequent dives to 5-20 m during daytime and 30-40 m during nighttime, with durations for nighttime dives running 6-7 minutes (Oleson et al. 2010). Some prey, such as mahi-mahi, occur most prevalently in the top 100 m of the water column, while

others, such as tuna and swordfish, may occur down to several hundred meters (Boggs 1992; Carey and Robinson 1981).

Feeding. HIFKWs are unique within their taxon as they are the only known group to exclusively exploit a shallow, productive coastal habitat versus ranging through oligotrophic waters, which may lead to the observation that HIFKWs have a relatively high density in nearshore Hawaiian waters versus false killer whales exploiting pelagic habitats (Acevedo-Gutierrez et al. 1997; Oleson et al. 2010). The primary prey of false killer whales are large pelagic fishes (Baird 2009; Baird et al. 2008; Baird et al. 1989; Brown et al. 1966; Bullis and Moore. 1956; Evans and Awbrey 1986; Kasuya 1985; Peacock 1936; Scheffer and Slipp 1948; Shallenberger 1981a; Silas et al. 1984; Tsutsumi et al. 1961), although marine mammals and squid may be predated upon (Baird et al. 1989; Bullis and Moore. 1956; Deraniyagala 1945; Hernandez-Garcia 2002; Hoyt 1983; Palacios and Mate 1996; Perryman and Foster. 1980; Rinaldi et al. 2007; Ross 1984). Some false killer whales shift diets seasonally (Tsutsumi et al. 1961). Few data are available to address which specific species HIFKWs target, but jacks, mahimahi, filefish, rainbow runner, amberjack, wahoo, tuna, marlin, moonfish, swordfish, lustrous pomfret, and others may be significant (Baird 2009; Baird et al. 2008; Brown et al. 1966; Oleson et al. 2010; Shallenberger 1981a). False killer whales have been known to remove large fishes on longlines (reports indicate tuna of 50-100 kg and one marlin >227 kg), leaving only the heads (Yuen 1977; Zimmerman 1983) and have been observed to capture free-swimming, highly-evasive mahimahi estimated at 8-9 kg (Brown et al. 1966). Attacks on large, highly mobile fishes, such as yellowfin tuna and broadbill swordfish have also been observed (Baird et al. 2008).

Feeding likely occurs cooperatively (Oleson et al. 2010) and prey sharing also has been documented (Baird et al. 2008; Connor and Norris 1982). Foraging occurs throughout the day and night (Baird et al. 2008; Evans and Awbrey 1986). Energetic requirements from captive individuals (probably less energetically demanding than free-ranging individuals) has been found to range between 2.9-6.1% of body weight daily (Baird et al. 2009; Kastelein et al. 2000; Sergeant 1969; Van Dyke and Ridgway 1977).

Habitat. Habitats that HIFKWs may occur in include a wide range of depths (<50 to >4,000 m) (Baird et al. 2010). Movement patterns suggest individual-based island preferences for periods of days followed by wide-ranging movements to short-term residencies in other locations, possibly in association with prey density and movement (Baird 2009).

Status and trends. The HIFKW was proposed for listing as endangered on November 17, 2010 (75 FR 70169). No historical levels of HIFKW population size are known. Estimates based upon assumed biological parameters have suggested possible historical levels of 769-2,461 individuals (Oleson et al. 2010). Data from 1993-1998 support a population estimate of 121 individuals, which is likely negatively biased (Mobley Jr. et al. 2000; Oleson et al. 2010). The best available estimate of population size is 123 individuals, but this estimate is somewhat dated (Baird et al. 2005a). It is not known whether two groups of false killer whales who have not been seen to associate with insular false killer whales are a part of the population or part of a separate population. Current estimates of population size are 151 individuals without these groups and 170 with them (Oleson et al. 2010).

Aerial survey data suggest that the population has been in decline since at least 1989. During this year, three groups were seen near Hawaii (outside of the known range of any population except

the insular population) numbering an estimated 380, 460, and 470 individuals, respectively (Reeves et al. 2009a). Aerial surveys since this time through 2003 have encountered gradually fewer individuals (Baird 2009; Mobley 2004; Mobley Jr. et al. 2000). Resighting rates have also been low during this time. Findings of surveys are supported by genetic analyses, which suggest a recent population decline (Chivers et al. 2010).

Natural threats. Reduced genetic diversity may be a natural, but partially anthropogenically induced factor leading to HIFKW decline (Oleson et al. 2010). Only a single instance of depredation on false killer whales has been documented, where killer whales attacked, killed, and consumed a false killer whale calf off New Zealand (Heithaus 2001; Visser et al. 2010). Parasitic infections have risen to levels thought to contribute to the deaths of some false killer whales, but these were from stranded individuals and it is unknown whether other health issues allowed for unhealthy levels of parasitism to develop (Andrade et al. 2001; Hernandez-Garcia 2002; Morimitsu et al. 1987; Odell et al. 1980; Sedlak-Weinstein 1991; Stacey et al. 1994; Zylber et al. 2002).

Anthropogenic threats. Several threats have been identified that may have or continue to lead to the decline of HIFKWs. These include competition with fisheries for prey, bioaccumulation of contaminants, live captures for aquaria, and injury from longline fisheries (Oleson et al. 2010). False killer whales in Hawaiian waters have been seen to take catches from longline and trolling lines (Nitta and Henderson 1993; Shallenberger et al. 1981). Interactions with longline and troll fishery operations appear to result in disfigurement to dorsal fins, with roughly 4% of the population showing this injury, as well as entanglement and hooking (Baird and Gorgone 2005; Forney and Kobayashi. 2007; McCracken and Forney 2010; Nitta and Henderson 1993; Shallenberger et al. 1981; Zimmerman 1983). Carretta et al. (2009) estimated that 7.4 individuals per year are killed or seriously injured during the course of fishing operations in the Hawaiian EEZ. In this area, false killer whales are the most frequently hooked or entangled cetacean species, with most interactions occurring in tuna-targeting longline operations (Forney and Kobayashi. 2007; McCracken and Forney 2010). In total, 31 observations of serious injury or mortality have been documented from 1994-2008, which has led to an estimated 13 false killer whales killed or seriously injured throughout the Hawaiian longline fishery (Forney and Kobayashi. 2007; McCracken and Forney 2010). It is noteworthy that most interactions occurred well beyond the range known for HIFKWs (0.6 HIFKWs were estimated to have been killed or serious injured from 2003-2008)(McCracken and Forney 2010). In addition, false killer whales depredate on catches from shortline fisheries at least off northern Maui, with deliberate shootings occurring in some cases (Nitta and Henderson 1993; NMFS 2009a; Schlais 1985; TEC 2009).

Overfishing of some pelagic fishes, including bigeye and yellowfin tuna, may be adversely affecting HIFKWs. Catch weights for mahimahi have also declined since 1987 (NMFS 2009c). These changes may limit the prey quantity or quality available for HIFKWs.

Bioaccumulation of particularly organic contaminants may be more of a concern for false killer whales than for many other cetaceans due to the high trophic level at which false killer whales feed. The only available study of HIFKW contaminant burden found PCBs and DDT present, with adult females carrying lower burdens than subadults or adult males (likely due to contaminants being unloaded into fetuses and milk during lactation) (Aguilar and Borrell. 1994; Krahn et al. 2009b; Ylitalo et al. 2009). PCB levels were high enough that biological effects

would be experienced in other mammals (Kannan et al. 2000). Persistent organic pollutant levels are similar between false killer whales sampled in Taiwan and Japan, but smaller (some much smaller) than samples from British Columbia (Chou et al. 2004; Haraguchi et al. 2006; Ylitalo et al. 2009). Although these pollutants are believed to typically be sequestered in blubber, individuals undergoing metabolic stress mobilize fat tissue, resulting in pollutants being mobilized into other body tissues (Aguilar et al. 1999). False killer whales from Australia and Japan have been found to have relatively high body burdens of mercury, lead, and cadmium (Endo et al. 2010; Kemper et al. 1994).

Bearded seal-Beringia and Okhotsk DPSs

Description of the species. Two subspecies of bearded seals are recognized by the NMFS: *Erignathus barbatus nauticus* in the Pacific and *Erignathus barbatus barbatus* in the Atlantic (Burns 1981; Davis et al. 2008b; Manning 1974; Rice 1998c). The two subspecies are delineated at 112º W and 145º E longitude (Cameron et al. 2010). Only *Erignathus barbatus nauticus* has been proposed for listing, in two segments: the Beringia and Okhotsk DPSs.

Distribution. Bearded seals proposed for listing in the Pacific and Arctic oceans are distributed from 85º N south to Sakhalin Island (45º N), including the Bering and Okhotsk Seas (Allen 1880b; Heptner et al. 1976a; King 1983; Manning 1974; Ognev 1935; Scheffer 1958a). Bearded seals from the two DPSs have not been documented through satellite telemetry or other means to move between DPS regions (Cameron et al. 2010).

Growth and reproduction. Singleton pups and rarely twins are born on ice flows from March to May with a peak in April (Burns 1981; Cameron et al. 2010; Heptner et al. 1976a; Johnson 1966; Kovacs et al. 1996; Krylov et al. 1964a; Reeves et al. 1992). Females give birth away from conspecifics (Heptner et al. 1976a; Kovacs et al. 1996). Pups are led away from the birthing flow soon after birth (Lydersen and Kovacs. 1999). Pups enter the water within hours of birth (Burns 1967; Gjertz et al. 2000b; Holsvik 1998; Kovacs et al. 1996; Tikhomirov 1966). *Erignathus barbatus nauticus* pups are weaned by 24 days, during which time females are mostly in water (split evenly between diving/foraging and activities near the surface) and pups are hauled-out, diving, or active near the surface (Burns 1981; Holsvik 1998; Krafft et al. 2000; Lydersen et al. 1994; Lydersen et al. 1996; Lydersen and Kovacs. 1999; Watanabe et al. 2009). Mothers nurse one to four times daily (usually between mid-morning and late-evening) and travel an average of 48 km per day (Krafft et al. 2000). During this time, mothers lose weight rapidly (4.5 kg/day) (Andersen et al. 1999; Heptner et al. 1976a; Krafft et al. 2000; Krylov et al. 1964a; Lydersen and Kovacs. 1999). Pups appear to accompany mothers on foraging dives and traveling (Gjertz et al. 2000b; Hammill et al. 1994; Lydersen et al. 1994; Lydersen and Kovacs. 1999; Watanabe et al. 2009). Stomach contents analyses support independent foraging prior to weaning (Burns 1981; Chapskii 1938). Growth during the first two years is rapid, with standard length increasing 9- 26% in the first year and 12-15% in the second, with growth slowing to 4% by the fourth year (Burns 1981; Burns and Frost 1983). Seasonal changes occur in body tissues, with individuals being leanest during summer after molting; the best available data indicate females lose roughly 9% of their body weight between late-winter and summer, while males lose much larger more (37%), with significance decreases in blubber thickness (Burns 1967; Burns 1981; Burns and Frost 1983). Females can appear larger than males in some areas, but measurements have thus far not resulted in significant differences (Benjaminsen 1973; Burns

1981; Chapskii 1938).

Aspects of breeding are almost entirely unknown, as significant portions of breeding and copulation likely occur underwater out of view of researchers (Cameron et al. 2010; Stirling 1983). Mating may be monogamous or promiscuous (Budelsky 1992; Stirling 1983; Stirling and Thomas. 2003). Adult females ovulate after lactation (Atkinson 1997; McLaren 1958b; Potelov 1975a). Examination of ovaries indicate ovulation occurs from March to June, with a peak in April to May (Burns 1967; McLaren 1958b; Potelov 1975a; Tikhomirov 1966). Breeding behavior is thought to occur during the parturition/nursing period. Following fertilization, implantation of the embryo is delayed by 2-2.5 months until between June and mid-August (Burns 1967; Burns 1981; Chapskii 1938; McLaren 1958b). Gestation lasts roughly nine months (Burns 1967).

Adult males engage in extensive underwater calling (apparently the only sex that vocalizes underwater) and territory defense (Burns 1967; Burns 1981; Cameron et al. 2010; Chapskii 1938; Cleator et al. 1989; Cleator and Stirling. 1990; Davies et al. 2006; Dubrovskii 1937; Freuchen 1935; Poulter 1968; Ray et al. 1969; Risch et al. 2007; Stirling et al. 1983; Van Parijs and Clark. 2006; Van Parijs et al. 2001a; Van Parijs et al. 2003a; Van Parijs et al. 2004). Calling peaks after weaning (Wollebaeck 1927). Males appear to defend single territories of less than 12 km^2 within and between years, although roaming over larger areas while calling has also been documented (Cleator et al. 1989; Cleator and Stirling. 1990; Risch et al. 2007; Van Parijs et al. 2003b; Van Parijs et al. 2004). Use of one or the other strategies is thought to be largely influenced by sea ice conditions, with areas having more predictable sea ice conditions associated with more territorial males (Burns 1967; Van Parijs 2003; Van Parijs et al. 2004). Breeding season in the Bering Sea is generally in May and June (Braham et al. 1981; Fiscus et al. 1976).

Migration and movements. Bearded seals move northward in late-spring and summer and southward during fall in association with preferred sea ice conditions (Burns 1967; Burns 1981; Burns and Frost. 1979; Fay 1974; Frost et al. 2008; Heptner et al. 1976a; Johnson 1966; Nelson 1981; Potelov 1969; Simpkins et al. 2003). Although these conditions are dynamic (particularly in recent years with dramatic changes in Arctic ice conditions), adults in the Bering Sea likely move through the Bering Strait during spring where they summer along the edge of the Bering and Chukchi Sea pack ice and multiyear ice (Burns 1981; Burns and Frost. 1979; Fay 1974; Heptner et al. 1976a; Nelson et al. 1984). Other individuals (mostly juveniles) stay closer to shore, including bays, brackish water estuaries, river mouths, and even up rivers during summer and early-fall (Burns 1967; Burns 1981; Heptner et al. 1976a). As ice reforms during fall and winter, seals remain with the edge of the ice and move back south through the Bering Strait, although this movement is less predictable than the northward migration (Burns 1981; Burns and Frost. 1979; Cameron and Boveng 2007; Cameron and Boveng 2009; Frost et al. 2005; Frost et al. 2008; Kelly 1988a). The central and northern Bering Sea appear to have the highest density of bearded seals at this time (Burns 1981; Burns and Frost. 1979; Fay 1974; Heptner et al. 1976a; Nelson et al. 1984)(Braham et al. 1981). Late-winter and early-spring is a time of diffuse occurrence for bearded seals, with individuals ranging along the ice front in the Chukchi and Bering Seas; young-of-the-year seals are particularly widely traveled, occurring as far south as the southern Kamchatka Peninsula (Burns 1967; Burns and Frost. 1979; Cameron 2006; Frost et al. 2008). However, individuals appear to maintain residence in their same general locations for weeks to months, likely exploiting local feeding areas (Cameron and Boveng 2009; Cameron

2005). In the Sea of Okhotsk, seasonal movements and preferences are similar to those previously described, but are of shorter distance (Fedoseev 1966; Fedoseev 2000b; Heptner et al. 1976a). In areas where sea ice completely disappears, some individuals haul-out on land in late summer and early-fall until sea ice reappears along the coast; this is most common in the Sea of Okhotsk, where tens to hundreds of individuals may occupy rookeries (Burns 1981; Chugunkov 1970; Heptner et al. 1976a; Krylov et al. 1964a; Tavrovskii 1971; Tikhomirov 1961b). Haul-outs are often low, pebbly banks or sandy shoals of bays and river mouths (Heptner et al. 1976a). Seal residency in nearshore waters versus following the receding ice may be influenced by ice receding into waters that are too deep for effective foraging or complete disappearance of sea ice (Burns 1981; Burns and Frost. 1979).

Vocalization and hearing. Males are the more vocal of the sexes, producing underwater frequency-modulated calls up to 1 min in duration that can be heard over 30 km away generally associated with breeding or display (Cleator et al. 1989; Van Parijs 2003; Van Parijs and Clark. 2006; Van Parijs et al. 2001b; Van Parijs et al. 2003b; Van Parijs et al. 2004).

Behavior. Limited observations of free-ranging bearded seals have been conducted and those that are available characterize relatively small areas and few individuals (Cameron et al. 2010). Individuals in the Sea of Okhotsk appear to haul-out more frequently in the afternoon and evening (late-May to July time frame), while individuals in the Bering and Chukchi Seas and Kotzebue Sound, Alaska rarely hauled-out at all (July to April), suggesting a lack of sea ice as necessary habitat during much of the year (Cameron et al. 2010; Frost et al. 2008; Heptner et al. 1976a). However, younger individuals may haul-out later in the day (evening) than adults (afternoon)(Frost et al. 2008). Mothers tolerate a greater level of threat to remain with or reunite with pups than other ice seals (Burns and Frost. 1979; Hammill et al. 1994; Krylov et al. 1964a; Lydersen et al. 1994).

Most diving data stem from nursing mothers and pups (Cameron et al. 2010). Diving behavior largely reflects local bathymetry, with relatively shallow (<100 m), short dives (<10 min) that are likely associated with bottom feeding (Gjertz et al. 2000b; Krafft et al. 2000). Pups appear to be capable of diving to the depths of adults within six weeks (Gjertz et al. 2000b). Maximum dives can exceed 300 m and pups can apparently dive to nearly 500 m (Cameron and Boveng 2009; Gjertz et al. 2000b; Kovacs 2002).

Bearded seals are capable of making and maintaining holes to enter, exit, and breath in areas of ice where breaks are not otherwise available (Burns 1967; Burns 1981; Burns and Frost. 1979; Cleator and Smith 1984; Fay 1974; Fedoseev 1965a; Nelson et al. 1984; Smith 1981b).

Feeding. Bearded seals feed on a wide variety of species, including benthic invertebrates, demersal fishes, and occasionally, schooling fishes (Antonelis et al. 1994; Burns 1967; Eschmeyer et al. 1983; Finley and Evans. 1983; Johnson 1966; Lowry et al. 1980b; Mecklenberg et al. 2002). Individuals likely switch prey to suit the available diet, sometimes on a seasonal and long-term basis (Dehn et al. 2007; Kosygin 1971). However, a few prey items seem to constitute the bulk of bearded seal diets, including bivalve mollusks, crabs, shrimp, sculpin, as well as Arctic, saffron, and polar cod (Allen 1880b; Antonelis et al. 1994; Bukhtiyarov 1990; Burns 1967; Burns and Frost. 1979; Chapskii 1938; Chugunkov 1970; Dehn et al. 2007; Dunbar 1941b; Fedoseev and Bukhtiyarov 1972; Finley and Evans. 1983; Frost et al. 1977; Heptner et al. 1976a; Hjelset et al. 1999; Johnson 1966; Kenyon 1962; Lowry et al. 1979; Lowry et al. 1980b; Nelson

et al. 1984; Nikolaev and Skalkin 1975; Ognev 1935; Pikharev 1941; Vibe 1950; Wilke 1954). In the Bering Sea and Sea of Okhotsk, crabs, shrimps, worms, and clams are the major prey items (Antonelis et al. 1994; Burns 1967; Burns and Frost. 1979; Frost et al. 1977; Kenyon 1962; Kosygin 1971; Lowry et al. 1979; Lowry et al. 1980b; Nelson et al. 1984). Juveniles appear to feed on more isopods, shrimp, and saffron cod than adults, while clams, crabs, sculpins, and flatfishes are of greater significance in the diets of older individuals (Burns and Frost. 1979; Lowry et al. 1980b; Nelson et al. 1984). Age-specific differences in feeding also include temporal differences, with younger animals feeding earlier in the day than older individuals (Frost et al. 2008; Kosygin 1971).

Bearded seals are thought to scan the bottom with their highly-innervated whiskers and burrow only in pursuit of prey (Marshall et al. 2006). Bearded seals use suction to draw the soft tissues of their mollusk prey from their shells (Burns 1981; Marshall et al. 2008).

Habitat. As bearded seals forage on benthic organisms year-round, they occur in waters where they can dive to the bottom (Burns 1981; Burns and Frost. 1979; Fedoseev 1984; Fedoseev 2000b; Heptner et al. 1976a; Kosygin 1971; Kovacs 2002; Nelson et al. 1984). Bearded seals generally prefer areas where they do not need to make and maintain holes in ice (Fedoseev 1984; Heptner et al. 1976a; Nelson et al. 1984). Aerial survey sightings in Pacific and Canadian waters suggest a preference for waters <100 m deep with open ice cover (Kingsley et al. 1985; Stirling et al. 1977; Stirling et al. 1982; Tarasevich 1963). Sexual segregation by depth may occur in some areas, with males occurring in areas with deeper water (Tarasevich 1963). Bearded seals are dependent upon sea ice for molting, pupping, and nursing, as well as for resting and thermoregulation, although they can occur on a variety of ice types and sizes (Burns 1981; Burns 2002; Burns and Frost. 1979; Fay 1974; Feltz and Fay. 1966; Lydersen and Kovacs. 1999; Nelson et al. 1984). They avoid unbroken, drifting, heavy ice as well as multi-year ice (Burns 1981; Burns and Frost. 1979; Burns and Harbo. 1977; Fedoseev 1965a; Fedoseev 1984; Kingsley et al. 1985; Nelson et al. 1984; Smith 1981b). In the Bering Sea, bearded seals were seen most frequently when ice coverage was greater than 75%; the same has been observed elsewhere (Kingsley et al. 1985; Simpkins et al. 2003; Ver Hoef et al. *In review*). Seals move with changing ice conditions, apparently maintaining this preference (Burns and Frost. 1979; Heptner et al. 1976a).

Status and trends. Beringia and Okhotsk DPSs of bearded seals were proposed for listing on December 10, 2010 (75 FR 77469). Bearded seals are born at a roughly even male to female sex ratio, but older age classes may exhibit a small bias towards females (Johnson 1966). Pup mortality has been found to be about 60% the first year, dropping to 19% by age 1, reaching lows of 8% from ages 6-20 before increasing with age (Burns and Frost. 1979; Nelson et al. 1984). Roughly 20% of pups survive to sexual maturity. Females reach sexual maturity at ages 5-6, with males following at an average of one year later (Andersen et al. 1999; Burns 1967; Burns and Frost. 1979; McLaren 1958b; Smith 1981b; Tikhomirov 1966). Although ovulation may occur as early as three years of age, pregnancy generally does not occur until age five (Cleator 1996; Fedoseev 2000b; Heptner et al. 1976a; Nelson et al. 1984; Potelov 1975b; Quakenbush et al. 2010; Smith 1981b; Tikhomirov 1966). Pregnancy rates for females six years old and older are 80% or higher (Burns 1967; Burns 1981; Burns and Eley 1978; Cleator 1996; Popov 1976; Potelov 1975b; Quakenbush et al. 2010; Smith 1981b; Tikhomirov 1966). The species is not known to become reproductively senescent (Burns and Frost. 1979; Cleator 1996). Bearded seals

live to an age of 20-25 years, with maximum longevity of 30 years (Andersen et al. 1999; Benjaminsen 1973; Burns and Frost. 1979; Kovacs 2002; Smith 1981b).

No accurate estimates of any bearded seal grouping are believed to exist due to issues in being able to accurately count seals (Cameron et al. 2010). However, estimates do exist. Overall, bearded seals of both subspecies have been estimated to number 500,000-1,000,000 combined by various authors (Blix 2005; Bychkov 1971; Kovacs 2002; Stirling and Archibald. 1979).

Bychkov's original estimate included roughly 450,000 individuals in the range of *Erignathus barbatus nauticus* (Bychkov 1971). Burns (1981) estimated 300,000 individuals in the Bering and Chukchi Seas a decade later. A variety of regional surveys in the Bering Sea have been conducted, but few data are available to estimate population size. The NMFS biological review team recommended setting current estimates at 125,000 for the Bering Sea and 27,000 for the Chukchi Sea based upon the most recent aerial surveys in these areas (Cameron et al. 2010).

Aerial surveys in the Sea of Okhotsk were the basis of original population estimates for this region, numbering 233,000-253,000 individuals (Fedoseev 1970; Fedoseev 1971). Fedsoseev (2000b) provided similar estimates using data collected from 1968-1990, with an estimated 200,000-250,000 individuals inhabiting the region. However, the NMFS biological review team recommended setting current estimates at 95,000 individuals based upon the last surveys conducted in the region (Cameron et al. 2010).

As population estimates are so imprecise, detecting changes in them over time is not quantitatively possible at this time (Cameron et al. 2010). Even qualitative estimates, such as querying local hunters in Alaska, have failed to detect any clear trends in population abundance (Cameron et al. 2010). Therefore, we have no means to determine whether bearded seal DPSs are declining, increasing, or stable. However, bearded seals are considered to still occupy the full extent of their range (Cameron et al. 2010)

Natural threats. Bearded seals are known to be predated upon by polar and brown bears, killer whales, and possibly walrus and Greenland sharks, with polar bears being the primary predator (Cameron et al. 2010; Cleator 1996; Fay 1960; Heptner et al. 1976a; Kelly 1988a; Lowry and Fay 1984; Lowry et al. 1987; Zenkovich 1938). Pups and subadults may be more vulnerable to polar bear predation than other age classes (Stirling and Archibald. 1977).

Anthropogenic threats. Both bearded seal DPSs were proposed for listing due to the potential impact that a warming climate may have on the biology of species, specifically the availability of ice and prey abundance and distribution, as well as possible impacts of ocean acidification on the marine food chain (Cameron et al. 2010).

Subsistence hunting of bearded seals has likely occurred for hundreds if not thousands of years in northern communities (Cameron et al. 2010; Krupnik 1984; Riewe 1991). Monitoring from 1966-1979 support annual harvests of roughly 1,700 individuals from the Bering, Chukchi, and East Siberian Seas, although a peak of 4,750-6,308 individuals occurred in 1977 (Burns 1981; Matthews 1978). From 1990-1998, levels had risen to a mean harvest of 6,788 individuals annually; these levels are believed to persist to the present day (Allen and Angliss 2010; Cameron et al. 2010; Coffing et al. 1998; Georgette et al. 1998; Wolfe and Hutchinson-Scarbrough 1999). However, correcting for individuals not recovered (bodies sink when in

water) results in estimates of 8,485-10,182 (Cameron et al. 2010). Subsistence hunting on the Chukchi Peninsula support annual average harvests of 629 individuals from 1941-1960 that declined through the 1960's and 1970's to an average harvest of 427 individuals, likely due to excessive harvest by commercial sealing and additional focus on walrus hunting (Cameron et al. 2010; Krupnik 1984). Total subsistence harvests from the Russian Bering and Chukchi Seas during the 1970's averaged 1,600 animals, with years possibly as high as 2,300, but decreasing to about 900 seals between 1981 and 1983 (Burns and Frost. 1979; Cameron et al. 2010; Fedoseev 2000b). Subsistence harvest levels in the Sea of Okhotsk are generally unknown (Cameron et al. 2010) .

Beginning in 1961, Soviet commercial sealing in the Bering and Chukchi Seas began, peaking in the late 1960's at over 7,000 individuals killed annually, after which levels dropped in response to overexploitation (Burns and Frost. 1979; Krylov et al. 1964a; Popov 1976; Popov 1982a). This sealing was suspended from 1970-1975. Commercial exploitation continued during 1981-1983 at far lower levels (average of 719 individuals)(Mineev 1984). Commercial harvests of bearded seals in the Sea of Okhotsk began in the early 1940's and quickly reached 6,000 individuals killed annually (Cameron et al. 2010). In 1957, a peak of 13,229 individuals was reached after the depletion of eastern Sakhalin Island and a shift to Shelikov Bay (Popov 1976). Harvests continued to fluctuate between 5,000 and 9,000 individuals for several years as exploited areas were depleted and hunts moved to new territory (Fedoseev 1984). From 1969-1980, harvests dropped dramatically to an average of 1,334. Commercial exploitation is now regulated and is set at 6,250 individuals for the Russian Bering and Chukchi Seas, Chukotka Peninsula, and eastern Siberia and 2,100 for the Sea of Okhotsk (Russian Federal Fisheries Agency 2009). Significantly, reported numbers were not corrected for struck and lost animals; depending upon hunting technique, estimates are likely 20-70% higher (Burns 1967; Davis et al. 1980; Fedoseev 2000b; Riewe and Amsden 1979; Smith and Taylor. 1977).

Organochlorine compounds have been found in bearded seals throughout their range and appear to be ubiquitous in the Arctic marine food chain (Bang et al. 2001b; Burrell 1981; Clausen 1978; Galster and Burns 1972; Galster 1971; Hoekstra et al. 2003; Kovacs 2007b; Muir et al. 2003; Norstrom and Muir. 1994; O'Hara and Becker. 2003; Quakenbush et al. 2010; Quakenbush and Citta. 2008; Smith and Armstrong. 1978; Wiberg et al. 2000). Levels of DDT in bearded seals from the Beaufort and Bering seas were higher than other Arctic marine mammals, although PCB levels were similar between Arctic pinnipeds (Galster and Burns 1972; Kelly 1988a)

Low bycatch and mortality (up to four individuals) has been reported for the Bering Sea/Aleutian Islands (BSAI) groundfish trawl fishery (Allen and Angliss 2010; Angliss and Allen 2009; Angliss and Lodge 2002). Bottom trawling also causes significant changes to the benthic environment which can impact bearded seals and the prey they rely upon (Cameron et al. 2010).

Ringed seal-Arctic, Baltic, Ladoga, and Okhotsk DPSs and Saimaa seal

Description of the species. Ringed seals may consist of up to ten subspecies based upon skull morphology, coat coloration, behavior, and genetics, but the NMFS currently recognizes five (Arctic, Baltic, Ladoga, Okhotsk, and Saimaa) with the understanding that additional information which is currently lacking may find additional classifications within the Arctic subspecies (Allen 1880a; Amano et al. 2002; Anderson 1934; Chapskii 1955; Davis et al. 2008a; Fedoseev 1984;

Hyvärinen and Nieminen 1990; Kelly et al. 2010b; Kelly et al. 2009b; King 1983; Ognev 1935; Palo 2003; Rice 1998b; Scheffer 1958b; Sell 2008).

Distribution. Ringed seal occurrence is circumpolar as well as in three lakes as remnants from glacial periods (King 1983). Arctic ringed seals occur as far south as Newfoundland and Baffin Bay in the Atlantic and the Bering Sea in the Pacific (King 1983; Mansfield 1967). Sea of Okhotsk ringed seals occur as far south as the Sea of Japan (King 1983). Arctic ringed seals do not come ashore, but rely entirely upon ice as a substrate for nursing, resting, and cover (Kelly 1988b; Kelly et al. 2010a). In areas where ice disappears entirely (all other subspecies), land is used for some of these functions (Härkönen et al. 1998; Kunnasranta 2001; Lukin et al. 2006; Ognev 1935; Trukhin 2000)

Growth and reproduction. For all subspecies, parturition occurs in late-winter to early-spring (February-April) in subnivean lairs (Arctic, Baltic, Ladoga, and Saimaa subspecies) or shoreline ice (lake subspecies) during maximal snow depth; Sea of Okhotsk pups are born in moving pack ice either in lairs or in the open (Belikov and Boltunov 1998; Chapskii 1940; Fedoseev 1965a; Fedoseev 1972; Fedoseev 1975; Finley and Evans 1983; Freuchen 1935; Helle 1979; Heptner et al. 1976b; Kelly 1988b; Krylov et al. 1964b; Kunnasranta 2001; Kunnasranta et al. 2001; Lukin et al. 2006; Lukin and Potelov 1978; Lydersen 1998; McLaren 1958a; Ognev 1935; Rautio et al. 2009; Sinisalo et al. 2008; Sipilä and Hyvärinen 1998; Smith 1987; Tikhomirov 1961a). Nursing continues for an average of 39 days post partum, but can vary from 3-9 weeks (Fedoseev 1975; Hammill et al. 1991; Käkelä and Hyvärinen 1993; Lydersen and Hammill 1993b). Pups spend about half of their time in water during the nursing period, diving up to 89 m deep and for as long as 12 minutes (Lydersen and Hammill 1993b). Just after weaning, pups shed their fetal coat for an adult-type coat (Kelly 1988b; Lydersen and Hammill 1993b). For all individuals, molting occurs from mid-May to mid-July with some regional variation in timing; individuals spend long periods out of the water and metabolism decreases by nearly 20% (Ashwell-Erickson et al. 1986; Kelly et al. 2010a; Kelly and Quakenbush 1990; Kunnasranta et al. 2002; McLaren 1958a; Smith 1973; Smith and Hammill. 1981). However, molting can be deferred until August if suitable ice is not available (Bychkov 1965; McLaren 1958a). Sexual maturity occurs at 4-8 years of age for females and 5-7 years for males in the Arctic and Sea of Okhotsk, although individual body condition and population structure can influence the timing (Burns and Fay 1970; Frost and Lowry 1981; Holst et al. 1999; Kelly 1988b; Kovacs 2007a; Lydersen and Gjertz 1987; Mansfield 1967; McLaren 1958a; Reeves 1998; Sipilä 2003; Sipilä and Hyvärinen 1998; Sipila et al. 1999; Smith 1973; Smith and Stirling 1975; Tikhomirov 1968). Females and males may mature more quickly in the Baltic (5 and 3-4 years, respectively), Lake Ladoga (4-5 and 6-7 years, respectively), and Lake Saimaa (4-5 and 4 years, respectively)(Curry-Lindahl 1975; McLaren 1958a; Popov 1979; Sipilä and Hyvärinen 1998). Pregnancy or ovulation rates in the Arctic have been found to vary between 0.45 and 0.86, although later revisions eliminating young individuals reduced much of the variability, with averages between 0.63 and 0.81 in various locations (Hammill 1987; Johnson et al. 1966; Nazarenko 1965; Reeves 1998; Smith 1987; Smith et al. 1979). However, Baltic levels were much lower (0.28), apparently due to a uterine pathology unique to the region (Helle 1980a; Reeves 1998). Lake Saimaa ringed seals have been reported to have pregnancy rates of between 0.7 and 0.83 between the 1980s and 1990s (Sipilä 2003; Sipilä et al. 1990). Ringed seals live to between 15 and 28 years of age on average, with maximum lifespan measured at 48 years (Frost and Lowry 1981; Helle 1980a; Holst et al. 1999; Lydersen and Gjertz 1987; McLaren 1958a; Sipilä and Hyvärinen 1998; Sipila et al. 1999; Smith

1973). Data from Russian harvests in the Sea of Okhotsk between 1960-1965 suggest a majority of individuals were less than 10 years old (Fedoseev 1968). Mortality rates derived from harvest data suggest a mortality rate of 30-41% for pups, dropping to 10% annually by sexual maturity and slowly increasing after age 15 (Kelly 1988b). Body condition changes drastically with season, with extensive blubber loss during spring and early summer due to reduced foraging, molting, and increased involvement with breeding and/or rearing of young (Fedoseev 1965b; Hammill et al. 1991; Johnson et al. 1966; Kelly et al. 2010b; Lowry et al. 1980a; Lydersen 1995; Lydersen and Hammill 1993a; Lydersen and Kovacs 1999; McLaren 1958a; Pikharev 1946; Ryg et al. 1990; Ryg and Øritsland 1991; Smith 1987). Females have been found to lose 19% of their body weight between March and June while males lost 12% (Ryg et al. 1990). These body reserves are replaced during the rest of the year (Kelly et al. 2010b).

The ringed seal mating system is believed to polygamous, with males defending territories they mark with a strong scent (particularly around breathing holes and adjacent snow) (Chapskii 1940; Hardy et al. 1991; Kelly et al. 2010a; Kelly et al. 2010b; Ognev 1935; Ryg et al. 1992; Smith 1981a; Smith 1987; Stirling 1977). Males in the Arctic rut from late-March to mid-May, with regional peaks in activity (Bakulina 1989; McLaren 1958a). Adult and subadult males appear to have bite marks and engage in aggressive behavior during the breeding season, a time when underwater vocalizations are documented to increase (Rautio et al. 2009; Smith 1987; Smith and Hammill. 1981; Stirling et al. 1983). Males may guard territories or mates underneath the sea ice, based upon interpretations of shallow dive depths and restricted movements of males versus females during the breeding season (Kelly et al. 2010a; Kelly et al. 2010b; Kelly and Wartzok 1996; Rautio et al. 2009; Stirling 1973; Stirling et al. 1983). Although size does not appear to correlate to the number of female neighbors, male age does and may influence reproductive success for individual males (Krafft et al. 2007). Mating has not been observed to date, but is thought to occur underwater near the females' lair (Kelly 1988b; Kelly et al. 2010b). Arctic females ovulate in May and early-June shortly after parturition, although ovulation can be suppressed if body condition is insufficient (Harwood et al. 2000; Johnson et al. 1966; Kelly et al. 2010b; Smith 1973; Smith 1987). Implantation is delayed by 3-3.5 months, followed by an approximate eight-month gestation for a single pup or, rarely, twins (Fedoseev 1975; McLaren 1958a; Smith 1987). Births occur at a 1:1 sex ratio (Fedoseev 1975; Frost and Lowry 1981; Helle 1980a; Lydersen and Gjertz 1987; McLaren 1958a; Sipilä et al. 1990; Sipila et al. 1999; Smith 1973).

Behavior. Arctic ringed seals are strongly driven by ice cover, with a typical year broken-up into three "ecological seasons": August to October as an open water or feeding period when intensive feeding occurs, an early-winter to March or May period when seals are resting in subsurface caves, and a breeding/molting period once ice begins to melt and break-up (Born et al. 2004; Kelly et al. 2010a) (Kelly et al. 2010b).

Arctic ringed seals in the Beaufort and Chukchi Seas spend most of their time either in water or in snowy lairs (90% August-November, 20% December-March), except during the spring molt (May-June) when they spend an average of 55% of their time basking on ice (Kelly et al. 2010a; Smith and Stirling 1975). Similar general observations have occurred in Baffin Bay and the Baltic Sea (Born et al. 2002; Harkonen et al. 2008; Teilmann et al. 1999). Arctic ringed seals rest in their lairs from April to mid-May (mostly at night)(Kelly et al. 2010a). Ringed seals spend more time on ice once spring temperatures warm and lairs start becoming exposed (late-

March to early April in the Sea of Okhotsk, late-April in Lake Ladoga, March to early June in the Bering and Chukchi Seas)(Heptner et al. 1976b; Kelly and Quakenbush 1990; Kunnasranta et al. 2002; Lowry et al. 1980a; Tikhomirov 1961a). Basking while molting reaches a peak in the Arctic during June (Born et al. 2002; Carlens et al. 2006; Harwood et al. 2007; Kelly et al. 2010a; Moulton et al. 2002; Smith 1987; Smith and Hammill. 1981). Individuals frequently return to the water, with pups entering and exiting more frequently than adults (Carlens et al. 2006). However, time out of water increases in June (Kelly et al. 2010a). When hauled out, individuals are vigilant and oriented for quick reentry into the breathing hole and/or facing downwind (Finley 1979; Kingsley and Stirling 1991). As sea ice breaks up, individuals spend more time in water (Kelly et al. 2010b).

Ringed seals are able to dive to depths in excess of 500 m for 39 minutes or more, although most dives are less than 10 minutes in duration and extend to whatever depth the ocean bottom is (Born et al. 2004; Gjertz et al. 2000a; Harkonen et al. 2008; Kelly and Wartzok 1996; Kunnasranta et al. 2002; Lydersen 1991; Teilmann et al. 1999). Diving ability improves with body size (Kelly 1997; Kelly and Wartzok 1996; Teilmann et al. 1999). Diving and resting patterns appear to be seasonally influenced, with more time spent out of water during the day and diving at night from spring to early-summer (breeding and molting) and the opposite true at all other times (Carlens et al. 2006; Kelly et al. 2010a; Kelly and Quakenbush 1990; Kunnasranta et al. 2002; Lydersen 1991; Teilmann et al. 1999).

Migration and movements. Movements can be most wide-ranging during the "open water" period from summer to fall, with individuals potentially ranging several hundred kilometers in the Arctic subspecies; some individuals may undergo much more limited movement (Bailey and Hendee 1926; Gjertz et al. 2000a; Harkonen et al. 2008; Harwood and Smith. 2003; Heide-Jørgensen et al. 1992; Kapel et al. 1998; Kelly and Wartzok 1996; Smith 1976; Smith et al. 1973; Smith and Stirling 1978; Teilmann et al. 1999). Following the period of open water foraging, adults of Arctic, Ladoga, and Saimaa subspecies return to the same areas they came from the previous winter (Kelly et al. 2010a; Koskela et al. 2002; Krafft et al. 2007; Kunnasranta et al. 2001; Sipilä et al. 1996; Smith and Hammill. 1981). Movements are more limited in late-fall and winter, ranging over just a few square kilometers unless they have access to leads in ice, in which case individuals can range over thousands of square kilometers (Born et al. 2004; Harwood et al. 2007; Kelly et al. 2010a; Kelly and Quakenbush 1990). As temperatures warm and snow melts in late-spring and early-summer, ice remains largely intact but seals spend extensive time basking in the sun during the molt (Finley 1979; Kelly et al. 2010a; Smith 1973). As Arctic individuals complete molting, they spend more and more time in the water (Kelly et al. 2010a).

Feeding. Ringed seals are catholic feeders predating at several trophic levels with regional and seasonal differences (Bradstreet and Cross 1982; Bradstreet and Finley 1983; Chapskii 1940; Dehn et al. 2007; Dunbar 1941a; Fedoseev 1965b; Gjertz and Lydersen 1986; Holst et al. 2001; Johnson et al. 1966; Labansen et al. 2007; Lowry et al. 1980a; McLaren 1958a; Nazarenko 1967; Pikharev 1946; Siegstad et al. 1998; Smith 1987; Söderberg 1971; Stenman and Pöyhönen 2005; Tormosov and Rezvov 1978; Wathne et al. 2000; Węsławski et al. 1994). However, preferred prey tend to be dense, schooling small fishes, although larger prey may be eaten (Kovacs 2007a; Smith 1977). Arctic ringed seals prey preferentially on gadid fishes (particularly Arctic, polar, and saffron cod as well as navaga) from late-autumn to early-spring (Belikov and Boltunov 1998; Bradstreet and Finley 1983; Chapskii 1940; Fedoseev 1965b; Gjertz and Lydersen 1986; Holst et

al. 2001; Johnson et al. 1966; Kovacs 2007a; Labansen et al. 2007; Lowry et al. 1980a; McLaren 1958a; Siegstad et al. 1998; Smith 1987; Wathne et al. 2000; Węsławski et al. 1994). Other regionally significant prey species include sculpins, capelin, redfish, snailfish, pricklebacks, and sand lance (Johnson et al. 1966; Labansen et al. 2007; Siegstad et al. 1998; Vincent-Chambellant 2010; Węsławski et al. 1994). Sea of Okhotsk ringed seas also prey extensively on herring and smelt during this time (Fedoseev 1965b). Baltic Sea ringed seals feed on herring, smelt, sculpins, Atlantic salmon, sticklebacks, and ruffe (Sinisalo et al. 2008; Sinisalo et al. 2006; Söderberg 1971; Tormosov and Rezvov 1978).

During the open water period, invertebrates tend to constitute more of an individual's diet, particularly of younger animals (Bradstreet and Finley 1983; Dunbar 1941a; Fedoseev 1965b; Holst et al. 2001; Johnson et al. 1966; Lowry et al. 1980a; Siegstad et al. 1998; Smith 1987; Węsławski et al. 1994). Prey include large amphipods, mysids, euphausiids, shrimps, and squids (Bradstreet and Finley 1983; Chapskii 1940; Dehn et al. 2007; Dunbar 1941a; Fedoseev 1965b; Gjertz and Lydersen 1986; McLaren 1958a; Siegstad et al. 1998; Smith 1987; Söderberg 1971; Węsławski et al. 1994).

In freshwater, European perch and smelt, common roach, vendace, and ruffe are important prey species for Lake Saimaa ringed seals, with invertebrates being of lesser importance (Kunnasranta et al. 1999). Lake Ladoga ringed seals feed on smelt, vendace, ruffe, burbot, stickleback, perch, roach, European whitefish, brown trout, fourhorn sculpin, and occasionally river lamprey and crayfish (Tormosov and Filatov 1979).

No sex-based differences in prey choice are known to exist (Bradstreet and Finley 1983; Gjertz and Lydersen 1986; Holst et al. 2001; Johnson et al. 1966; Lowry et al. 1980a). However, younger seals may prefer crustaceans over fish, a trend seen to reverse in older individuals (Bradstreet and Cross 1982; Bradstreet and Finley 1983; Fedoseev 1965b; Lowry et al. 1980a; Siegstad et al. 1998; Smith 1987). Arctic ringed seals feed extensively upon cod just before waters freeze, likely to build-up energy reserves for the breeding, estrus, and lactation period (Harwood and Stirling 1992; Kelly et al. 2010b; Smith 1987).

Habitat. Ringed seals haul out on ice year-round to rest, although they may also use rocky reefs, islands, shorelines, and sand bars when ice is unavailable (Harkonen et al. 2008; Hyvärinen et al. 1995; Krylov et al. 1964b; Lukin et al. 2006; Sipilä et al. 1996). Ringed seals are particularly adept at scrapping and clawing breathing holes (even in heavy winter ice up to two meters thick) as well as subnivean (within snow pack) lairs over these holes (Bailey and Hendee 1926; Hammill and Smith 1989; Kelly 1996; Kelly et al. 2010b; Lukin and Potelov 1978; Ognev 1935; Smith and Stirling 1975). As snow accumulates above holes, ringed seals excavate lairs for resting, nursing, thermoregulation, predator avoidance, and parturition (Bengtson et al. 2005; Burns 1970; Finley and Evans 1983; Hammill and Smith 1991; Kelly et al. 2010b; McLaren 1958a; Smith et al. 1991; Wiig et al. 1999). Snow depth is rarely sufficient over open ice on lakes Ladoga and Saimaa and occurs only along the shoreline, which is where ringed seal lairs are found (Sipilä and Hyvärinen 1998). Models of thermoregulation suggest that pups could not thermoregulate effectively in some areas without the thermal refuge that lairs provide (Kelly 1988b; Smith et al. 1991; Taugbøl 1982).

Status and trends. Arctic, Sea of Okhotsk, Baltic Sea and Lake Ladoga DPSs of ringed seals were proposed for listing as threatened on December 10, 2010 (75 FR 77476). The Siamaa seal

was listed as endangered on July 28, 1993 (58 FR 40538). As with other ice seals, data for estimating abundance and trends is extremely difficult to obtain and no comprehensive studies exist. Worldwide estimates have been suggested at several million individuals (Reeves 1998; Stirling and Calvert. 1979).

The Arctic subspecies, due to its wide distribution, is believed to be the most abundant subspecies of ringed seal. Estimates at various Greenland and Baffin Bay locations include: 200,000 near Svalbard (Jødestøl and Ugland 1994), 7,585 near Spitsbergen (Carlens et al. 2006), more than 28,000 in Kong Oscars Fjord, Scoresby Sound (Born et al. 1998) , 67,000 in northeastern along the shore of Baffin Bay and 417,000 within the pack ice (Finley and Evans 1983) , 97,800 for eastern Baffin Bay (Miller et al. 1982) , and 787,000 on pack ice of Canada and Greenland (Finley and Evans 1983). This last estimate is the only comprehensive estimate for the region and abundance has been suggested to be stable (Kelly et al. 2010b). Hudson Bay has also been surveyed frequently, with estimates including 455,000 in western Hudson Bay (Smith 1975), 280,000 for the same region a quarter century later (Lunn et al. 1997), 73,170 in 2007 and 33,701 in 2008 (Ferguson and Secretariat 2009). The BRT concluded that a mean between these two last estimates (53,436) was most reasonable; no estimate of trend is available (Kelly et al. 2010b). Early estimates of ringed seals in the Alaskan Beaufort Sea estimated 40,000 seals during the winter months (Burns and Harbo 1972). Bengtson et al. (2005) estimated 252,488 individuals in 1999 and 208,857 in 2000 for the Alaskan Chukchi Sea. Estimates of 250,000 individuals in the shorefast ice and 1-1.5 million individuals in the pack ice for the combined Beaufort and Chukchi Seas have been made (Frost 1985). An estimated 30,900 individuals occurred in the Amundsen Gulf in 1981 and 70,500 in 1982 (Kingsley and Lunn 1983). The BRT estimated that at least one million individuals inhabit the Beaufort and Chukchi Seas (Kelly et al. 2010b). Estimates from the White, Barents, Kara, and East Siberian Seas are generally lacking, although these areas encompass half of the Arctic subspecies' habitat, but some estimates have been put forth, the largest being 2-2.5 million for the eastern Barents Sea to the Bering Sea (Heptner et al. 1976b). Estimates for the Barents Sea include 35,000-50,000 individuals from 1988-1994 as well as 24,000-30,000 individuals in the White Sea from the 1970s-1980s (Ognetov 2002). The Kara Sea has been estimated to support 90,000-150,000 individuals (Ognetov 2002).

Early (1968-1970) estimates of the Sea of Okhotsk DPS suggest a population size of 800,000 individuals, including 130,000 around Sakhalin Island; both numbers were likely higher due to uncounted seals in the water (Fedoseev 1970; Fedoseev 1971; Shustov 1972). However, this is probably a good estimate for pre-commercial hunting abundance (Kelly et al. 2010b). More recent surveys in 1986 estimated 7,490 individuals (Lagarev 1988). Feoseev (2000) estimated 676,000-855,000 individuals in the Sea of Okhotsk between 1998 and 1990; the BRT estimates that the lower bound of this estimate is the best available current population size for the DPS (Kelly et al. 2010b).

The Baltic Sea may once have been home to 50,000-450,000 ringed seals in 1900 (Harding and Härkönen 1999; Kokko et al. 1999). However, by the 1940s, hunting had reduced population numbers to 25,000 individuals (Harkonen et al. 2008). Abundance appeared to have stabilized thereafter until 1965, when populations again declined, this time to 5,000 individuals by the late 1970s, due to organochlorine contamination and subsequent ringed seal sterility (Bergman and Olsson 1985; Harding and Härkönen 1999; Helle 1980b). The greatest concentration of ringed

seals occurred in Bothnian Bay, where 2,000-3,000 individuals were estimated to live in the 1970s, after which abundance declined through the mid-1980s (Härkönen and Lunneryd 1992; Helle 1990). About 2,500 individuals were estimated in counts from 1988-1990 in this region (Härkönen and Lunneryd 1992). Currently, about 75% of Baltic Sea ringed seals are believed to occur in Bothnian Bay (6,100 individuals) and growth rate here is estimated at 4.5% annually (Karlsson et al. 2008; Kelly et al. 2010b). In addition to Bothnian Bay, 149-173 individuals are currently believed to occur in the Gulf of Finland, where population growth is stable (Härkönen et al. 1998). The Gulf of Riga was estimated to have 1,407 individuals in 1996 (Härkönen et al. 1998). Härkönen et al. (1998) estimated 5,510 individuals for the entire Baltic Sea.

Lake Ladoga may once have hosted 20,000 individuals in the 1930s; this estimate decreased to between 5,000 and 10,000 by the 1960s (Kelly et al. 2010b; Zheglov and Chapskii 1974). Zheglov and Chapskii (1974) observed 2,200 seals and 5,100 holes in aerial surveys during the early 1970s. Another subsequent aerial survey during the same decade estimated 3,500-4,700 individuals. In 2001, aerial surveys estimated 2,000 individuals basking on the ice. The latest estimate of abundance for the population is 3,000-5,000 individuals (Agafonova et al. 2007).

Lake Saimaa is believed to have supported 4,000-6,000 individuals 5,000 years ago (Sipilä 2006; Sipilä and Hyvärinen 1998). However, in 1893, the population was estimated at 100-1,300 individuals (Kokko et al. 1999). Sipilä et al. (1990) estimated that the population declined from 250 individuals in 1971 to 140 in 1984. Currently, fewer than 300 individuals are believed to survive (Kelly et al. 2010b; Sipilä 2006). Trends suggest that the northern and southern regions of the lake have experienced significant declines in abundance, whereas the central portion has remained relatively stable (Kelly et al. 2010b). Mean growth rate suggests some recovery between 1990 and 2004 (Kelly et al. 2010b; Sipilä 2006).

Natural threats. Predators are the main natural threat of ringed seals and include polar and brown bears, Arctic and red foxes, gray wolves, lynx, European mink, walruses, killer whales, Greenland sharks, common ravens, and glaucous gulls (Burns and Eley 1976; Fay et al. 1990; Heptner et al. 1976b; Melnikov and Zagrebin 2005; Sipilä 2003). Ringed seals are one of the primary prey species for polar bears, with ringed seals composing 80-98% of polar bear diet in the Beaufort Sea and Hudson Bay region during some periods (Derocher et al. 2004; Heptner et al. 1976b; Stirling and Parkinson. 2006). From 8-44% of pup production may be removed by polar bear predation (Hammill and Smith 1991). Ringed seals are particularly vulnerable to predation from polar bears as they spend more time on ice molting as well as when lairs disintegrate earlier than expected, such as from rainfall or low snowfall (Hammill and Smith 1991; Messier et al. 1992; Stirling 1974). Early lair exposure can also expose pups to avian predation (Gjertz and Lydersen 1983; Kumlien 1879; Lydersen 1998; Lydersen and Gjertz 1987; Lydersen et al. 1987; Lydersen and Ryg 1990; Lydersen and Smith 1989). Along with polar bears, Arctic foxes can exert regionally high levels of predation on newborn pups from the Arctic ringed seal subspecies (Kelly and Quakenbush 1990; Kelly et al. 1986; Lydersen and Gjertz 1984; Smith 1976).

Anthropogenic threats. Arctic, Ladoga, Baltic, and Sea of Okhotsk DPSs were proposed due to the potential impact that a warming climate may have on the biology of the species, specifically the availability of ice and prey abundance and distribution, as well as possible impacts of ocean acidification on the marine food chain (Kelly et al. 2010b). As ringed seals rely upon lairs for

resting, nursing, thermoregulation, predator avoidance, and parturition, early spring break-ups can adversely impact growth, condition, and survival of pups (Harwood et al. 2000; Lukin et al. 2006; Stirling and Smith 2004). The ringed seal BRT expects early breakups to occur more frequently as a result of warming temperatures and adversely impact ringed seal productivity and abundance via pup survival (Ferguson et al. 2005; Kelly 2001; Kelly et al. 2010b; Smith and Hammill 1980; Stirling and Smith 2004). Prey distribution, particularly of Arctic cod, may also shift as a result of temperature changes (Kelly et al. 2010b).

Ringed seals have been hunted for subsistence for 1000s of years, a practice which continues presently (ACIA 2005; Hovelsrud et al. 2008; Kovacs 2007a; Krupnik 1988). Alaskan harvests killed 7,000-15,000 individuals annually from 1962-1972, but declined to 3,000-6,000 during 1973-1977 and 2,000-3,000 by 1979 (Frost 1985). Currently, 9,500 individuals are estimated to be harvested annually in Alaska (Allen and Angliss 2010). In Canada, harvests may have been as high as 100,000 individuals annually during the 1960s and 1970s, leading to local depletion (Mansfield 1970; Reeves et al. 1998b). Harvests declined through the 1980s and 1990s to the 50,000-65,000 range (60,000-80,000 accounting for killed and lost animals)(Reeves et al. 1998b). Greenland hunts likely took about 43,000 individuals during the 1950s, 45,000-75,000 animals annually in the 1960s, rising to near 100,000 individuals annually in the 1970s before declining to roughly 70,000 annually during the 1980s and 1990s (Teilmann and Kapel 1998). Catches from 1995-2006 averaged 82,421 animals and have included both subsistence and commercial components (Greenland Home Rule 2009). Harvests here focused on younger age classes. Russian catches numbered in the few thousands (up to 13,200) annually in the White and Barents Seas, respectively (Belikov and Boltunov 1998; Krupnik 1993; Nazarenko 1969; Timoshenko 1984). Most of these areas experienced declines in harvest between the 1960s and 1970s into the 1980s due to decreased commercial demand (Belikov and Boltunov 1998; Timoshenko 1984). A few thousand individuals were also harvested in the Russian Bering Sea between 1961 and 1969, which likely continued through 1990 (Fedoseev 2000a). By far the largest Russian harvests of ringed seals occurred in the Russian Bering and Chukchi Seas by subsistence hunters. Native harvests are estimated at 25,000 in the late 1930s, 23,500 by the 1940s, and 15,500 by the late 1950s (Heptner et al. 1976b). Harvests along the Bering Sea were 30,000-35,000 after World War II, but decreased to 10,000-12,000 annually (Popov 1982b). Fedoseev (1984) estimated the combined harvest along the Bering, Chukchi, and East Siberian Seas was 40,000 individuals between 1940 and 1954. However, shore-based harvests have been restricted to 2,000-3,000 individuals since 1970 (Popov 1982b). Harvests reportedly numbered 991-3,607 individuals along the Bering and Chukchi Seas between 1979 and 1983 (Mineev 1981; Mineev 1984). The decline in harvests was likely due to native peoples shifting to a modern lifestyle (Fedoseev 1984). Sea of Okhotsk harvests likely depleted the population, with subsistence harvests amounting to 25,000-30,000 individuals before 1950 and increasing to an average of 70,000 individuals annually in the 1950s and 1960s, not accounting for killed but lost seals (Fedoseev 1984; Fedoseev 2000a; Heptner et al. 1976b; Krylov et al. 1964b; Popov 1982b). This extensive hunting likely altered the age structure of the population (Fedoseev 1984; Krylov et al. 1964b). Harvest regulation began in 1969, with levels decreasing to 32,742 that year and 25,000 in 1974 (Popov 1982b). Average harvests declined to 4,834 individuals per year from 1976-1980 (Fedoseev 1984). Ship-based harvests from 1990-1994 averaged 12,664 individuals (Grachev 2006).

Hunting has also been significant for other ringed seal subspecies. Hunting in the Baltic Sea was

conducted such that hundreds of thousands of individuals were reduced to 6,000-10,000 by the 1980s (Almkvist 1982; Helle 1986). Dramatic declines in Lake Ladoga were also caused by excessive hunting (Sipilä and Hyvärinen 1998). Here, 436-1,278 individuals were harvested annually between 1909 and 1918 (Heptner et al. 1976b). An additional 950 individuals were killed in the northern portion of the lake from 1924-1939 (Jääskeläinen 1942). From 10,000 to 20,000 individuals were killed as part of bounties from 1950 to 1974 (Agafonova et al. 2007; Verevkin et al. 2006). Hunting was banned shortly thereafter. Bounties were also given for Lake Saimaa ringed seals from 1882 to 1948 (Metsähallitus 2009). A total of 246 seals were killed from 1893 to 1905 and an additional 140 were shot from 1909 to 1918; 278 were killed between 1928 and 1947 (Kilkki and Marttinen 1984; Kokko et al. 1998). Since a ban on their killing in 1955, only two cases of poaching are known (Ashley et al. 2008; Sipilä 2003; Sipilä and Hyvärinen 1998; Wilson et al. 2001).

A variety of contaminants have been identified in ringed seals, some to the point of causing sterility. Organic contaminants have also been identified in ringed seals, including DDT, DDE, and PCBs (Addison et al. 2005; Addison and Smith 1974; Bang et al. 2001a; Helle et al. 1983; Helle et al. 1976a; Helle et al. 1976b; Helle and Stenman 1984; Kostamo et al. 2000; Kucklick et al. 2002; Nakata et al. 1998; Nyman et al. 2002; Riget et al. 2006; Sipilä and Hyvärinen 1998). Levels in the Baltic Sea were correlated with ringed seal reproductive failure (Helle et al. 1976a; Helle et al. 1976c). Perflourinated compounds have also been identified in ringed seals, with little understanding of their significance (Bossi et al. 2005; Butt et al. 2007; Kannan et al. 2002; Kannan et al. 2001; Martin et al. 2004; Quakenbush and Citta. 2008).
Heavy metals, including mercury, cadmium, lead, selenium, arsenic, zinc, chromium and nickel have been found to accumulate in ringed seal liver and kidney (Atwell et al. 1998; Gaden et al. 2009; Helle 1981; Hyvärinen et al. 1998; Koeman et al. 1975; Quakenbush and Sheffield 2007; Riget et al. 2005; Smith and Armstrong 1978; Sonne et al. 2009; Wagemann 1985; Wagemann 1989; Wagemann et al. 1996). Mercury and selenium accumulate with age (Dietz et al. 1998; Helle 1981; Hyvärinen et al. 1998; Medvedev et al. 1997; Riget et al. 2005; Smith and Armstrong 1978). Cadmium peaked at 5-10 years of age and declined thereafter (Dietz et al. 1998). Mercury has been found to be higher in Baltic females than males (Helle 1981). Nickel might play a role in stillborn pup mortality (Hyvärinen and Sipilä 1984).

Fisheries bycatch is a significant issue for Baltic Sea, Lake Saimaa, and Lake Ladoga ringed seals. An estimated 80 seals are bycaught in nets annually along the Swedish coast (Lunneryd et al. 2005). Of the 182 carcasses examined from 1977 to 2000, more than half were determined to have died from drowning in fishing nets (Sipilä 2003). Measures to reduce bycatch likely resulted in a decrease mortality rate of 1-3 individuals annually by the late 1990s (Sipilä and Hyvärinen 1998). Annual mortality of Lake Ladoga ringed seals was estimated at 200-400 individuals as a result of fisheries interaction in the late 1980s and early 1990s; however, changes in the fishing industry have resulting in much reduced bycatch here (Sipilä and Hyvärinen 1998; Sipilä et al. 2002).

Critical habitat. Critical habitat has not been proposed for ringed seals.

Environmental Baseline

By regulation, environmental baselines for Opinions include the past and present impacts of all state, federal, or private actions and other human activities in the action area, the anticipated

impacts of all proposed federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of state or private actions that are contemporaneous with the consultation in process (50 CFR §402.02). The *Environmental baseline* for this Opinion includes the effects of several activities affecting the survival and recovery of ESA-listed whale species in the action area.

Climate change

In general, based on forecasts made by the Intergovernmental Panel on Climate Change (IPCC), climate change is projected to have substantial effects on individuals, populations, species, and the structure and function of marine, coastal, and terrestrial ecosystems in the near future (IPCC 2000; IPCC 2001a; IPCC 2001b; IPCC 2002). From 1906 to 2006, global surface temperatures have risen 0.74º C and continue to rise at an accelerating pace; 11 or the 12 warmest years on record since 1850 have occurred since 1995 and the past decade has been the warmest in instrumental history (Arndt et al. 2010; Poloczanska et al. 2009). Furthermore, the Northern Hemisphere (where a greater proportion of ESA-listed species occur) is warming faster than the Southern Hemisphere, although land temperatures are rising more rapidly than over the oceans (Poloczanska et al. 2009). Climate change will result in increases in atmospheric temperatures, changes in sea surface temperatures, patterns of precipitation, and sea level. Sea levels have risen an average of 1.7 mm/year over the 20th century and 3.3 mm/year between 1993 and 2006 due to glacial melting and thermal expansion of ocean water; this rate will likely increase, which is supported by the latest data from 2009 (Arndt et al. 2010; Hoegh-Guldberg and Bruno 2010; Wilkinson and Souter 2008). Oceanographic models project a weakening of the thermohaline circulation resulting in a reduction of heat transport into high latitudes of Europe, an increase in the mass of the Antarctic ice sheet, and a decrease in the Greenland ice sheet, although the magnitude of these changes remain unknown. Reductions in ozone and subsequent increases in ultraviolet radiation have been linked to possible skin damage and blistering in blue, fin, and sperm whales in the Gulf of California (Martinez-Levasseur et al. 2010).

Climate change has been linked to changing ocean currents as well. Rising carbon dioxide levels have been identified as a reason for a poleward shift in the Eastern Australian Current, shifting warm waters into the Tasman Sea and altering biotic features of the area (Poloczanska et al. 2009). Similarly, the Kuroshio Current in the western North Pacific (an important foraging area for juvenile sea turtles and other listed species) has shifted southward as a result of altered long-term wind patterns over the Pacific Ocean (Poloczanska et al. 2009).

Climate change would result in changes in the distribution of temperatures suitable for whale calving and rearing, the distribution and abundance of prey, and abundance of competitors or predators. For species that undergo long migrations, individual movements are usually associated with prey availability or habitat suitability. If either is disrupted by changing ocean temperature regimes, the timing of migration can change or negatively impact population sustainability (Simmonds and Eliott. 2009). Climate change can influence reproductive success by altering prey availability, as evidenced by high survival of northern elephant seal pups during El Niño periods, when cooler, more productive waters are associated with higher first-year pup survival (McMahon and Burton. 2005). Reduced prey availability resulting from increased sea temperatures has also been suggested to explain reductions in Antarctic fur seal pup and harbor porpoise survival (Forcada et al. 2005; Macleod et al. 2007). Primary production is estimated to

have declined by 6% between the early 1980s and 2010 partly as a result of climactic shifts, making foraging more difficult for marine species (Hoegh-Guldberg and Bruno 2010). Polygamous marine mammal mating systems can also be perturbated by rainfall levels, with the most competitive grey seal males being more successful in wetter years than in drier ones (Twiss et al. 2007). Sperm whale females were observed to have lower rates of conception following unusually warm sea surface temperature periods (Whitehead 1997). Marine mammals with restricted distributions linked to water temperature may be particularly exposed to range restriction (Issac 2009; Learmonth et al. 2006). MacLeod (2009) estimated that, based upon expected shifts in water temperature, 88% of cetaceans would be affected by climate change, 47% would be negatively affected, and 21% would be put at risk of extinction. Of greatest concern are cetaceans with ranges limited to non-tropical waters and preferences for shelf habitats, such as North Atlantic right whales (Macleod 2009). Variations in the recruitment of krill and the reproductive success of krill predators correlate to variations in sea-surface temperatures and the extent of sea-ice cover age during winter months. Although the IPCC (2001b) did not detect significant changes in the extent of Antarctic sea-ice using satellite measurements, Curran et al. (2003) analyzed ice-core samples from 1841 to 1995 and concluded Antarctic sea ice cover had declined by about 20% since the 1950s.

Foraging is not the only potential aspect that climate change could influence. Acevedo-Whitehouse and Duffus (2009) proposed that the rapidity of environmental changes, such as those resulting from global warming, can harm immunocompetence and reproductive parameters in wildlife to the detriment of population viability and persistence. Altered ranges can also result in the spread of novel diseases to new areas via shifts in host ranges (Simmonds and Eliott. 2009). It has been suggested that increases in harmful algal blooms could be a result of increases in sea surface temperature (Simmonds and Eliott. 2009). Warming temperatures are forecasted to open the Northwest Passage to shipping, introducing large amounts of shipping noise and potential for ship strike to arctic and subarctic regions that presently experience little vessel traffic (Alter et al. 2010).

Species that are shorter-lived, have larger body sizes, or are generalist in nature are liable to be better able to adapt to climate change over the long term versus those that are longer-lived, smaller-sized, or rely upon specialized habitats (Brashares 2003; Cardillo 2003; Cardillo et al. 2005; Issac 2009; Purvis et al. 2000). Climate change is likely to have its most pronounced effects on species whose populations are already in tenuous positions (Isaac 2008). As such, we expect the risk of extinction to listed species to rise with the degree of climate shift associated with global warming.

Naturally-occurring climatic shifts, such as the Pacific Decadal Oscillation, El Niño, and La Niña can strongly influence marine productivity, including marine mammals and the prey they rely upon (Beamish et al. 1999; Benson and Trites. 2002; Francis et al. 1998; Hare et al. 1999; Mantua et al. 1997). Cooler periods appear to promote coastal biological productivity in the action area and warmer phases have the opposite effect (Hare et al. 1999; NMFS 2008e). Changes in ocean temperature also directly influence salmon abundance in the Strait of Juan de Fuca and the vicinity of the San Juan Islands. In years when ocean conditions are cooler than usual, the majority of sockeye salmon returning to the Fraser River do so via this route, but when warmer conditions prevail, migration patterns shift to the north through Johnstone Strait, altering the value of foraging habitat for southern resident killer whales from year-to-year (Groot and

Quinn 1987).

Habitat degradation

A number of factors may directly or indirectly affecting listed species in the action area by degrading habitat; perhaps most significant among them is anthropogenic noise in the ocean. Natural sources of ambient noise include: wind, waves, surf noise, precipitation, thunder, and biological noise from marine mammals, fishes, and crustaceans. Anthropogenic sources of ambient noise include: transportation and shipping traffic, dredging, construction activities, geophysical surveys, and sonars. In general, it has been asserted that ocean background noise levels have doubled every decade for the last six decades in some areas, primarily due to shipping traffic (IWC 2004b). The acoustic noise that commercial traffic contributes to the marine environment is a concern for listed species because it may impair communication between individuals (Hatch et al. 2008). Shipping and seismic noise generally dominates ambient noise at frequencies from 20 to 300 Hz (Andrew et al. 2002; Hildebrand 2009; Richardson et al. 1995c). Background noise has increased significantly in the past 50 years as a result of increasing vessel traffic, and particularly shipping, with increases of as much as 12 dB in low frequency ranges and 20 dB versus preindustrial periods (Hildebrand 2009; Jasny et al. 2005; McDonald et al. 2006; NRC 1994; NRC 2003b; NRC 2005; Richardson et al. 1995c). Over the past 50 years, the number of commercial vessels has tripled, carrying an estimated six times as much cargo (requiring larger, more powerful vessels) (Hildebrand 2009). Seismic signals also contribute significantly to the low frequency ambient sound field (Hildebrand 2009). Baleen whales may be more sensitive to sound at those low frequencies than are toothed whales. Dunlop et al. (2010a) found that humpback whales shifted from using vocal communication (which carries relatively large amounts of information) to surface-active communication (splashes; carry relatively little information) when low-frequency background noise increased due to increased sea state. Sonars and small vessels also contribute significantly to mid-frequency ranges (Hildebrand 2009).

Commercial shipping in the Gulf of Alaska is dominated by cargo transports, container freight, crude oil tankers, and barges. Military vessels, ferries, and other commercial and recreational fishing vessels also converge in the Gulf of Alaska. Two primary shipping lanes radiate from the Gulf of Alaska to Honolulu and San Francisco. Important Alaskan ports include Kodiak, Alaska's largest commercial fishing port, and Valdez, the southern terminus of the 1,300 km trans-Alaska pipeline. Additional minor ports are located throughout the region and include: Anchorage, Cordova, Homer, Kodiak, Nikiski, Seward, Whittier, and Yakutat.

The Puget Sound and nearby waters experience very high levels of vessel traffic from both commercial and recreational sources, producing the potential for ship strike, high ambient noise levels, and behavioral harassment of southern resident killer whales. Commercially, a quarter million vessels move within the Puget Sound region annually, with the Ports of Seattle and Tacoma combining to be the third largest port in the U.S. (www.washingtonports.org). These vessels include tankers, tugs, cargo containers, ferries, and a variety of other vessel types. Several cruise ships are also based out of Seattle. Recreationally, 244 marinas, nearly 40,000 moorage slips, and 331 boat launches are located within the Pacific Northwest, servicing 180,000 registered recreational vessels and countless vessels not requiring registration (WSDE 2006). Haro Strait, one of the regions primary shipping lanes, is frequently used by southern resident killer whales.

Several major ports occur further south along the U.S. west coast, including Portland, San Francisco, Los Angeles, Long Beach, and San Diego (DoT 2005). These ports service a wide variety of vessels, including cargo, tug and barges, small ships, liquid bulk, dry bulk, break bulk, intermodal (container, roll-on/roll-off, lighter aboard ship), ferry, tourist passenger vessels (sailboats, ferry, party-boat fishing, whale watching) and cruise ships. Long Beach is among the largest ports in the U.S., accounting for 6% of the total cargo entering the U.S., and increasing rapidly (growing 122% between 2003 and 2006) (DoT 2007a; DoT 2007b). Los Angeles is also the fifth largest cruise ship terminal in the U.S. A shipping lane runs along the U.S. west coast south to southern California and additional shipping lanes extend westward from San Francisco and near Santa Barbara Island.

The northeastern U.S. hosts some of the busiest commercial shipping lanes in the world, including those leading into Boston, Providence, Newark, and New York (MARAD 2011). In addition, the Boston cruise ship terminal is growing rapidly and is currently one of the busiest cruise ship ports in the U.S. {Massport, 2002 #6}. In addition to vessel traffic, marine construction activities occur in the Cape Cod area (liquefied natural gas terminal construction, pile driving, offshore wind farm construction, dredging, cable laying, drilling, and others) that contributes to local and regional background sound levels.

In-water construction activities (e.g., pile driving associated with shoreline projects) in both inland waters as well as coastal waters in the action area can produce sound levels sufficient to disturb marine mammals under some conditions. Pressure levels from 190-220 dB re 1 μPa were reported for piles of different sizes in a number of studies (NMFS 2006b). The majority of the sound energy associated with pile driving is in the low frequency range (<1,000 Hz) (Illingworth and Rodkin Inc. 2001; Illingworth and Rodkin Inc. 2004; Reyff 2003). Dredging operations also have the potential to emit sounds at levels that could disturb marine mammals. Depending on the type of dredge, peak sound pressure levels from 100 to 140 dB re 1 μPa were reported in one study (Clarke et al. 2003). As with pile driving, most of the sound energy associated with dredging is in the low-frequency range, <1000 Hz (Clarke et al. 2003).

Several measures have been adopted to reduce the sound pressure levels associated with in-water construction activities or prevent exposure of marine mammals to sound. For example, a six-inch block of wood placed between the pile and the impact hammer used in combination with a bubble curtain can reduce sound pressure levels by about 20 dB (NMFS 2008e). Alternatively, pile driving with vibratory hammers produces peak pressures that are about 17 dB lower than those generated by impact hammers (Nedwell and Edwards 2002). Other measures used in the action area to reduce the risk of disturbance from these activities include avoidance of in-water construction activities during times of year when marine mammals or listed salmon may be present; monitoring for marine mammals during construction activities; and maintenance of a buffer zone around the project area, within which sound-producing activities would be halted when marine mammals enter the zone (NMFS 2008e).

Marine features in the central and northeastern Pacific are also subject to degradation. The continental shelf off Oregon and Washington is cut by numerous submarine canyons, which tend to trap sediments and pollutants associated with discharges stemming from coastal development (Airamé et al. 2003). Seamounts are hotspots for marine biodiversity, particularly for large pelagic species (Morato et al. 2010). These areas are sensitive to fishery impacts due to the high level of endemism characteristic of this habitat. Species that inhabit seamounts tend to be long-

lived and do not move widely between seamounts, meaning that their recovery can be very slow (Johnston and Santillo 2004; Richer de Forges et al. 2000). As several listed species appear to be drawn to seamounts, apparently due to prey availability there, the deterioration of the habitat could have significant effects on listed species.

Oil spills could have a significant deleterious effect on marine mammals that are exposed to them. Exposure can occur via skin contact, ingestion of oil directly or through contaminated prey, or inspired while at the surface (Geraci 1990). This exposure could result in displacement of marine mammals from an impacted area or produce toxic effects. Perhaps the most famous shipwreck of all time occurred in the Gulf of Alaska when, in 1989, the *Exxon Valdez* released at least 11 million gallons of Alaskan crude oil into one of the largest and most productive estuaries in North America. The spill was the worst in U.S. history until the Deepwater Horizon event in 2010. The Alaska Department of Environmental Conservation estimated that 149 km of shoreline was heavily oiled and 459 km were at least lightly oiled. Oil spills, both small and large, occur widely along U.S. shores at refining and transfer facilities and extraction sites.

Ingestion of marine debris can have fatal consequences even for large whales. In 1989, a stranded sperm whale along the Mediterranean was found to have died from ingesting plastic that blocked its' digestive tract (Viale et al. 1992). A sperm whale examined in Iceland had a lethal disease thought to have been caused by the complete obstruction of the gut with plastic marine debris (Lambertsen 1990). The stomach contents of two sperm whales that stranded separately in California included extensive amounts of discarded fishing netting (NMFS 2009). A fifth individual from the Pacific was found to contain nylon netting in its stomach when it washed ashore in 2004 (NMFS 2009). Further incidents may occur but remain undocumented when carcasses do not strand. North Pacific sperm whales may be exposed to high levels of marine debris due to trash accumulation in the North Pacific Gyre, which is estimated to contain 90.7 million metric tons of marine debris (Marks and Howden 2008).

Entrapment/entanglement in fishing gear and shooting

Fisheries interactions are a significant problem for several marine mammal species and particularly so for humpback (Figure 6). Aside from the potential of entrapment and entanglement, there is also concern that many marine mammals that die from entanglement in commercial fishing gear tend to sink rather than strand ashore, thus making it difficult to accurately determine the frequency of such mortalities. Entanglement may also make whales more vulnerable to additional dangers, such as predation and ship strikes, by restricting agility and swimming speed. Between 1998 and 2005, observers identified 12 humpback whales injured or killed by fisheries off the U.S. west coast (NMFS, unpublished data). An estimated 78 rorquals were killed annually in the offshore southern California drift gillnet fishery during the 1980s (Heyning and Lewis. 1990). From 1996-2000, 22 humpback whales of the Central North Pacific population were found entangled in fishing gear (Angliss and Lodge. 2004). In 1996, a vessel from the Pacific Missile Range Facility in Hawaii rescued an entangled humpback, removing two crabpot floats from the whale.

Figure 6. A humpback whale entangled in fisheries gear off Hawaii (Credit: NOAA, Hawaiian Islands Humpback Whale National Marine Sanctuary, ESA permit number 932-1489).

There are no observer program records of bowhead whale mortality incidental to commercial fisheries in Alaska (Angliss and Allen. 2009). However, some bowhead whales have historically had interactions with crab pot gear and there are several documented cases of bowheads having ropes or rope scars (Angliss and Allen. 2009). NMFS Alaska Region stranding reports document three bowhead whale entanglements between 2001 and 2005 (Angliss and Allen. 2009). In 2003, a bowhead whale was found dead in Bristol Bay entangled in line around the peduncle and both flippers; the origin of the line is unknown (Angliss and Allen. 2009). In 2004, a bowhead near Point Barrow was observed with fishing net and line around the head (Angliss and Allen. 2009).

Recent reports of entanglement are unknown, but Sheffer and Slipp (1948) documented several deaths of killer whales caught in gillnets between 1929 and 1943 in Washington State waters. Typically, killer whales are able to avoid nets by swimming around or underneath them (Jacobsen 1986; Matkin 1994). Recreational fishing also has the potential to affect fish habitats because of the large number of participants and the intense, concentrated use of specific habitats. Historically, killer whales have commonly been subject to shooting (some likely fatal) by fisherman due to perceived competition for target fish resources (Baird 2001b; Haley 1970; Olesiuk et al. 1990c; Pike and Macaskie. 1969; Scheffer and Slipp. 1948). This practice has largely abated in the past few decades and unlikely to continue today (Carretta et al. 2001; Young

et al. 1993).

Sperm whales are known to have been incidentally taken in drift gillnet operations, which killed or seriously injured an average of nine sperm whales annually from 1991-1995 (Barlow et al. 1997). Sperm whales have been bycaught in pelagic drift gillnets along the U.S. east coast and in artisanal gillnets targeting sharks and large pelagic fishes off the Pacific coasts of northwestern South America, Central America, and Mexico (Palacios and Gerrodette 1996; Waring et al. 1997). An individual was caught and released from gillnetting, although injured, on Georges Bank during 1990. A second individual was freed, but injured, from gillnetting on George's Bank in 1995. In 1994, a sperm whale was disentangled from gillnet along the coast of Maine. Interactions between longline fisheries and sperm whales have been common over the past decade (Rice 1989; Hill and DeMaster 1999). Between 1994 and 2002, one sperm whale was observed entangled within the Hawaiian Islands EEZ in the Hawaii-based longline fishery and was able to free itself without injury (Forney 2004). In August 1993, a dead sperm whale, with longline gear wound tightly around the jaw, was found floating ~32 km off Maine.

Acoustic harassment devices

Acoustic harassment devices (AHDs) are another source of underwater sound that may occur in the action area and may be disruptive to southern resident killer whales. AHDs used at salmon aquaculture farms emit "loud" signals intended to displace harbor seals and sea lions and thereby reduce depredation (NMFS 2008e; Petras 2003). However, these signals can also cause strong avoidance responses in cetaceans (Olesiuk et al. 2002). Morton and Symonds (2002) describe one AHD model that broadcasts a 10 kHz signal at 194 dB re 1 μPa at 1 m. A large majority of these occur in Arctic waters during exploration for petroleum products, although other detectable above ambient levels in open water for up to 50 km. Activation of AHDs at an aquaculture farm near northeastern Vancouver Island corresponded with drastic declines in the presence and use of nearby passages and inlets by both resident and transient killer whales (Morton and Symonds 2002). The only AHD still in use in Washington State operates at the Ballard locks in Seattle, where NMFS uses it to deter sea lions (NMFS 2008e).

Seismic surveys and oil and gas production

Numerous surveys have been conducted in the northeast Pacific, Arctic, and northwest Atlantic using seismic airguns, and have the potential to affect ESA-listed seismic surveys. As a general mitigation measure, airguns are shutdown if marine mammals approach too closely, presumably avoiding the potential for temporary or permanent threshold shifts in their hearing. However, some species (such as bowhead whales) appear to be particularly sensitive to seismic, vessel, and industrial sound sounds and may move rapidly away from the area at up to several kilometers from the sound source (Gallagher and Hall. 1993; George 2010; Greene 1982; Richardson et al. 1995a; Richardson et al. 1985a; Richardson et al. 1990; Richardson et al. 2004b; Richardson and Williams 2003; Richardson and Williams 2004; Schick and Urban 2000; Streever et al. 2008; Wartzok et al. 1989). Other baleen whales frequently do the same (Malme et al. 1984a; Malme et al. 1985; McCauley et al. 2000; McCauley et al. 1998a; McCauley et al. 1998b; Miller et al. 1999a; Stone and Tasker 2006).

The major sources of industrial underwater noise n the Arctic appear to be offshore oil, gas or mineral exploration and exploitation. These activities increase vessel traffic, produce loud

sounds for seismic profiling, place structures in areas used by whales, and introduce noises from drilling and production into the environment (NMFS 1999; NMFS 2006f). Malme et al. (1985) exposed feeding humpback whales in southeastern Alaska to noise from a single air gun or to playback of recorded sounds of oil drilling, production platforms and aircraft. Whales showed no overall pattern of avoidance during 13 experiments, each of which included between 10 and 40 different animals. Whales startled as soon as the airgun was turned on in three experiments. These startle responses, which occurred at received sound levels between 150 to 169 dB (re 1 mPa), were thought to be caused more by the novelty of the air gun sound than by its intensity. Oil and gas exploration and development are increasingly active in the Chukchi and Beaufort Sea inhabited by Western Arctic bowhead whales. There are presently two offshore production facilities within state waters in the Beaufort Sea: Northstar and Endicott. Multiple marine geophysical (seismic) projects are planned for the Beaufort and Chukchi Seas.

Naval activities

Naval activity, notably sonar use during training exercises, has gained notoriety for its coincidence with marine mammal strandings. However, other activities (also during training exercises in designated naval operating areas and training ranges) also have the potential to adversely impact marine mammals. The action area overlaps several naval training ranges or facilities listed below. Listed individuals travel widely in the North Pacific and could be exposed to naval activities in several ranges.

- Marianas Island Range Complex, where humpback and sei whales may or likely breed and give birth,
- The Okinawa and Japan Range Complexes,
- The Southern California Range Complex, where blue whales forage,
- The Northwest Training Range Complex, where humpback whales forage and southern resident killer whales reside,
- The Gulf of Alaska Operating Area, where several listed whale species are known to forage and Steller sea lions reside, and
- The Hawaiian Islands Operating Area, where humpback whales regularly breed and give birth.

Naval activities to which individuals could be exposed include, among others, vessel and aircraft transects, munition detonations, and sonar use. Responses by marine mammals could include no response, short-term and long-term behavioral responses and changes (altered vocal activity, changes in swimming speed and direction, respiration rates, dive times, and social interactions), temporary or permanent hearing loss, debris ingestion, ship-strike injury, and death. Death or injury is not expected to occur as a result of exposure to naval activities. Several unusual incidents of stranding or milling have occurred in association with naval activities on the Hawaii Range complex, but such incidents from other training ranges have not been documented.

Although naval vessels represent a small fraction of the total sound level and are designed to

operate quietly, these ships are large and equipped with high-output sonar equipment such as ANISQS-53C tactical sonar, which produces signals at source levels of 235 dB re 1 μPa_{rms} at 1 m. The signals emitted from these devices have the potential to affect marine mammals in the action area; however, empirical data are limited. An event that occurred in the Strait of Juan de Fuca and Haro Strait on May 5, 2003 demonstrates the potential for naval activities to impact southern resident killer whales. The U.S. Navy guided missile destroyer U.S.S. *Shoup* passed through the strait operating its mid-frequency sonar during a training exercise. Members of J pod (a family group of southern resident killer whales) were in the strait at the same time and exhibited unusual behaviors coincident with exposure to the sonar, as reported by local researchers (Commander U.S. Pacific Fleet 2003; NMFS 2005a; NMFS 2006b). Based on the duration of exposure, the received levels experienced by the whales, and information on sound levels known to cause behavioral reactions in other cetaceans, NMFS concluded J pod was exposed to levels likely to cause behavioral disturbance, but not temporary or permanent hearing loss (NMFS 2005a; NMFS 2006b). Underwater detonations are sometimes performed at this site and there was an occasion when J pod was less than 1.5 km away when a blast occurred, which caused the whales to suddenly change their direction of travel (NMFS 2006b). No stranding or mortality events have been documented in or around other operating areas or training ranges within the action area that appear linked to naval sonar, although five beaked whales were discovered stranded or floating dead coincident in time with the Alaska Shield/Northern Edge 2004 exercise between June 17-19, 2004 in the Gulf of Alaska Operating Area. However, no mid-frequency sonar or explosives were used during this exercise and evidence linking the exercise to mortalities is circumstantial at best.

Subsistence and commercial harvest

Eskimos have hunted bowhead whales along the coastlines of the Bering, Chukchi, and Beaufort Seas for at least 2,000 years (Stoker and Krupnik. 1993). Bowheads are harvested by the Inupiat people in the Alaskan Beaufort, Bering, and Chukchi Seas. In Alaska, 10 Native villages currently participate in subsistence whaling. Rice (1964) estimated an annual average of 10 bowhead whales killed and recovered each year, while another 3-4 whales were struck but lost for each one landed (40-50 strikes annually). The total Alaskan subsistence harvest of bowheads since 1973 and 1991 ranged from 17 to 111, although a limit of 67 strikes has been set since 1998 unless strikes from a previous year had not been used (up to 5 strikes per year in Russian waters and 77 by Alaska natives) (NMFS 2008b). The average take of Western Arctic bowhead whales by Alaska Natives over the last ten years, not including those struck and lost, was 41.8 (Suydam et al. 2007). The harvest consists mostly of immature whales, approximately evenly distributed between sexes (Philo et al. 1993).

Although the IWC protected sperm whales from commercial harvest in 1981, Japanese whalers continued to hunt sperm whales in the North Pacific until 1988 (Reeves and Whitehead 1997). In 2000, the Japanese Whaling Association announced plans to kill 10 sperm whales in the Pacific Ocean for research. Although consequences of these deaths are unclear, the paucity of population data, uncertainly regarding recovery from whaling, and re-establishment of active programs for whale harvesting pose risks for the recovery and survival of this species. Sperm whales are also hunted for subsistence purposes by whalers from Lamalera, Indonesia, where a traditional whaling industry has been reported to take up to 56 sperm whales per year. Japan also kills up to 101 sei whales annually (IWC 2008).

Subsistence hunting of Cook Inlet beluga whales has occurred since at least the 1700s and may occur in the future (Braund and Huntington. 2010). In 1999 and 2000 there was a voluntary moratorium on subsistence harvest (Carter and Nielsen. 2010), and since substance harvest have been conducted under co-management agreements. Since 2000, no more than 2 beluga whales have been taken in subsistence harvests in any one year (Hobbs et al. 2008). Take levels have been as high as 20% in the 1990s, but hunts are now tightly regulated and based upon population dynamics.

Vessel approaches – commercial and private marine mammal watching

Although considered by many to be a non-consumptive use of marine mammals with economic, recreational, educational and scientific benefits, marine mammal watching is not without potential negative impacts. Whale watching has the potential to harass whales by altering feeding, breeding, and social behavior or even injure them if the vessel gets too close or strikes the whale. Another concern is that preferred habitats may be abandoned if disturbance levels are too high. In the Notice of Availability of Revised Whale Watch Guidelines for Vessel Operations in the Northeastern United States (64 FR 29270; June 1, 1999), NMFS noted that whale watch vessel operators seek out areas where whales concentrate, which has led to numbers of vessels congregating around groups of whales, increasing the potential for harassment, injury, or even the death of these animals. In addition to whale watching vessels, large cruise vessels also operate in waters off the coast of Alaska, and may pose a threat to humpback whales. Whale watching, particularly of humpback whales, is extensive in Hawaiian waters during winter. The interactions that individuals experience in these waters likely influence how they react to approaches by vessels in the future (Herman 1979).

Several studies have specifically examined the effects of whale watching on marine mammals, and investigators have observed a variety of short-term responses from animals, ranging from no apparent response to changes in vocalizations, duration of time spent at the surface, swimming speed, swimming angle or direction, respiration rate, dive time, feeding behavior, and social behavior (NMFS 2006b). Responses appear to be dependent on factors such as vessel proximity, speed, and direction, as well as the number of vessels in the vicinity (Au and Green. 2000; Corkeron 1995; Erbe 2002b; Magalhaes et al. 2002; Richter et al. 2003; Scheidat et al. 2004; Watkins 1986; Williams et al. 2002b; Williams et al. 2002d). Foote et al. (2004b) reported that southern resident killer whale call duration in the presence of whale watching boats increased by 10-15% between 1989-1992 and 2001-2003 and suggested this indicated compensation for a noisier environment. Disturbance by whale watch vessels has also been noted to cause newborn calves to separate briefly from their mothers' sides, which leads to greater energy expenditures by the calves (NMFS 2006b). Although numerous short-term behavioral responses to whale watching vessels are documented, little information is available on whether long-term negative effects result from whale watching (NMFS 2006b).

It is difficult to precisely quantify or estimate the magnitude of the risks posed to marine mammals in general and southern resident killer whales specifically (who possibly have the greatest exposure to whale watching activities of any listed marine mammal) by whale watching and recreational vessels (NMFS 2008e). Commercial whale watching in Washington State has increased dramatically from small scale operations during the late 1970s to early 1980s, to 13 vessels by 1988, and a total of 76 vessels (and over 500,000 people) in 2006 (Koski 2006a;

Koski 2007b; NMFS 2008e; Osborne 1991). Most companies belong to the Whale Watch Operators Association Northwest, which has established whale viewing guidelines for commercial operators (WWOANW 2007). Currently, over 50% of vessels involved with whale watching are commercially owned, with the San Juan Islands and adjacent area also attracting large numbers of private boaters for recreational activities such as opportunistic viewing of killer whales (Koski 2007b; NMFS 2008e). In addition, private floatplanes, helicopters, and small aircraft regularly take advantage of whale watching opportunities (MMMP 2002a). Weather conditions in the Pacific Ocean in winter limit whale watching during winter months and activity is greatest during summer (NMFS 2008e). From May to September 2005, an average of over 19 boats (up to 94) surrounded southern resident killer whales on a daily basis (Koski 2006a). In Washington State, southern resident killer whales are the primary target species, particularly in Haro Strait (Hoyt 2001; Hoyt 2002; NMFS 2008e).

The increase in whale watching traffic over the past two decades has resulted in increased exposure of southern resident killer whales to vessel traffic and sound emitted by it. Whale watching activities have the potential to affect southern resident killer whales in the action area, resulting in possible disturbance or displacement. Increasing anthropogenic sound levels in the Puget Sound region have been associated with increased call duration by southern resident killer whales when vessels are present (Erbe 2002b; Foote et al. 2004b). Vessels also appear to cause whales to alter their direction of travel (Williams et al. 2002b; Williams et al. 2002d). Furthermore, vessel presence has been linked to reduced foraging success and/or inhibiting foraging all together (Bain et al. 2006b; Williams et al. 2006). Based on a study in Johnstone Strait, British Columbia, northern resident killer whales decreased feeding behaviors significantly and increased time engaging in behaviors which required less energy such as resting and socializing (Williams et al. 2006).

Live-captures for aquaria

Killer whales have been displayed in aquaria worldwide since the early 1960s. For 15 years, killer whales were collected from the wild to populate display facilities; all but one individual came from Washington State or British Columbia until 1976, when local laws banned captures (Hoyt 1990; NMFS 2006b). During this time, from 275-307 killer whales were captured, of which 55 were sent to aquaria, 12-13 died, and 208-240 were released or escaped. Of the individuals captured and displayed or killed, 70% (47 or 48 individuals) were southern resident killer whales, including 17 immature males, 10 immature females, nine mature females, and seven or eight mature males; 15 individuals were from K pod, five from L, and one from J (Baird 2001b; NMFS 2006b; Olesiuk et al. 1990c). The selective removal of younger animals and males produced a skewed age and sex composition in the southern resident killer whale DPS, which probably affected its ability to recover (Olesiuk et al. 1990c).

Ship-strike

Ship-strike is a significant concern for the recovery of baleen whales in the region. We believe the vast majority of ship-strike mortalities go unnoticed, and that actual mortality is higher than currently documented. More humpback whales are killed in collisions with ships than any other whale species except fin whales (Jensen and Silber 2003a). Along the Pacific U.S. coast, a humpback whale is known to be killed about every other year by ship-strikes (Barlow et al.

1997). Two whales have been struck offshore of Japan (Jensen and Silber 2003a).

Despite these reports, the magnitude of the risks commercial ship traffic poses to large whales in the proposed action areas has been difficult to quantify or estimate. We struggle to estimate the number of whales that are killed or seriously injured in ship strikes within the U.S. Exclusive Economic Zone and have virtually no information on interactions between ships and commercial vessels outside of U.S. waters. With the information available, we know those interactions occur but we cannot estimate their significance to whale species.

Shipstrike is also a concern for balaenopterids (Figure 7). In the California/Mexico stock of blue whales, annual incidental mortality due to ship strikes averaged one whale every 5 years, but we cannot determine if this reflects the actual number of blue whales struck and killed by ships (i.e., individuals not observed when struck and those who do not strand; Barlow et al. (1997)). The vast majority of ship strike mortalities are never identified, and that actual mortality is higher than currently documented. Jensen and Silber's (2004) review of the NMFS' ship strike database revealed fin whales as the most frequently confirmed victims of ship strikes (26% of the recorded ship strikes [n = 75/292 records]), with most collisions occurring off the east coast, followed by the west coast of the U.S. and Alaska/Hawaii. Five of seven fin whales stranded along Washington State and Oregon showed evidence of ship strike with incidence increasing since 2002 (Douglas et al. 2008). From 1994-1998, two fin whales were presumed killed by ship strikes. More recently, in 2002, three fin whales were struck and killed by vessels in the eastern North Pacific (Jensen and Silber 2003b). Ship strikes also present an emerging threat to sei and blue whales; in 2003, a sei whale was reported struck by a vessel, subsequently died, and stranded near Port Angeles, Washington (NMFS, unpublished data), and a blue whale was struck and killed off the coast of California in 2002 (Jensen and Silber 2003b).

Among baleen whales, ship strikes are least common for bowhead whales, although they do occur. Between 1976 and 1992, only three ship strike injuries were documented out of a total of 236 bowhead whales examined from the Alaskan subsistence harvest (George et al. 1994). Since that publication, the applicants note that six additional whales have been noted with ship strike injuries (1995-2002) out of approximately 180 examined whales, indicating that the rate of ship strikes may have increased slightly in recent years. The low number of observed ship strike injuries suggests that bowheads either do not often encounter vessels or they avoid interactions with vessels. It is likely that an unknown number of unobserved and unreported mortalities may occur after ship strikes. However, given the steadily increasing population trend, the magnitude of this potential effect is likely to be small.

There have not been any recent documented ship strikes involving sperm whales in the eastern North Pacific, although there are a few records of ship strikes in the 1990s. Two whales described as "possibly sperm whales" are known to have died in U.S. Pacific waters in 1990 after being struck by vessels (Barlow et al. 1997). There is an anecdotal record from 1997 of a fishing vessel that struck a sperm whale in southern Prince William Sound in Alaska, although the whale did not appear to be injured (Laist et al. 2001). More recently in the Pacific, two sperm whales were struck by a ship in 2005, but it is not known if these ship strikes resulted in injury or mortality (NMFS 2009b). The lack of recent evidence should not lead to the assumption that no mortality or injury from collisions with vessels occurs as carcasses that do not drift ashore may go unreported, and those that do strand may show no obvious signs of having been struck by a

ship (NMFS 2009b). Worldwide, sperm whales are known to have been struck 17 times out of a total record of 292 strikes of all large whales, 13 of which resulted in mortality (Jensen and Silber 2003a; Laist et al. 2001). Given the current number of reported cases of injury and mortality, it does not appear that ship strikes are a significant threat to sperm whales (Whitehead 2003).

Figure 7. A near collision between a blue whale and a commercial cargo vessel in the Santa Barbara Channel Traffic Separation Scheme. Photo credit: NOAA Channel Islands National Marine Sanctuary, 2002 (Permit CINMS-2002-001).

A total of six instances have been documented of northern and southern resident killer whales being struck by vessels since the 1990s, including lethal interactions (Baird 2001b; Carretta et al. 2001; Carretta et al. 2004b; Visser 1999; Visser and Fertl. 2000).

Scientific research and permits

Scientific research permits issued by the NMFS currently authorize studies of listed species in the Pacific and Atlantic Oceans, many of which extend into portions of the action area. Authorized research on ESA-listed whales includes close vessel and aerial approaches, biopsy sampling, tagging, ultrasound, and exposure to acoustic activities. Research activities involve non-lethal "takes" of these whales by harassment, with none resulting in mortality. Steller sea lions are exposed to approach, capture and restraint, biopsy, tagging, anesthesia or sedation, hot branding, lavage, ultrasound, blood or tissue sampling, tooth extraction, and authorized mortality.

Tables 8-19 describe the cumulative number of takes for each listed species in the action area authorized in scientific research permits.

Table 8. Blue whale takes in the North Pacific.

Year	Approach	Biopsy	Suction cup tagging	Implantable tagging	Acoustic playback
2009	12,430	1,065	328	105	0
2010	15,747	1,485	430	180	21
2011	8,327	525	385	120	21
2012	3,822	415	315	120	21
2013	3,322	415	225	75	21
Total	43,648	3,905	1,683	600	84

Permit numbers: 1127-1921, 1071-1770, 727-1915, 540-1811, 731-1774, 781-1824, 1058-1733, 774-1714, 782-1719, 808-1735, 14097, 14122, 14296, 14451, 14534, and 14585.

Table 9. Fin whale takes in the North Pacific.

Year	Approach	Biopsy	Suction cup tagging	Implantable tagging	Acoustic playback
2009	16,860	1,975	150	55	0
2010	21,273	2,975	410	130	80
2011	7,773	1,075	305	85	80
2012	4,468	1,015	265	75	80
2013	4,468	1,015	265	75	80
Total	54,842	8,055	1,395	420	320

Permit numbers: 1127-1921, 1071-1770, 473-1700, 540-1911, 731-1774, 781-1824, 782-1719, 1058-1733, 965-1821, 0642-1536, 1049-1718, 774-1714, 808-1735, 14097, 14122, 14296, 14451, and 14534.

Table 10. Sei whale takes in the North Pacific.

Year	Approach	Biopsy	Suction cup tagging	Implantable tagging
2009		435	75	25
2010	6,386*	730	250	100
2011	2,491	320	200	75
2012	2,271	310	180	75
2013	2,271	310	180	75
Total	17,589	2,105	885	350

Permit numbers: 1127-1921, 540-1811, 731-1774, 782-1719, 1058-1733, 1049-1718, 774-1714, 0642-1536, 808-1735, 14097, 14585, 14122, 14296, 14451, and 14534.

Table 11. North Pacific right whale takes in the North Pacific.

Year	Approach	Biopsy	Suction cup tagging	Implantable tagging	Acoustic playback
2009	717	88	42	42	0
2010	877	118	62	52	50
2011	425	68	60	50	50
2012	425	68	60	50	50
2013	278	68	24	14	50
Total	2,722	410	248	208	200

Permit numbers: 1058-1733, 14097, 782-1719, 14122, 14296, and 14585.

Table 12. Bowhead whale takes in the North Pacific.

Year	Approach	Biopsy	Suction cup tagging	Implantable tagging
2009	1,875	415	365	320
2010	2,070	530	430	385
2011	1,475	385	405	360
2012	975	135	155	110
2013	415	135	65	65
Total	6,810	1,600	1,420	1,240

Permit numbers: 1058-1733, 14097, 14610, 369-1757, 774-1714, and 782-1719.

Table 13. Humpback whale takes in the North Pacific.

Year	Approach	Biopsy	Suction cup tagging	Implantable tagging	Acoustic playback	Exhalation sampling	Ultrasound
2009	44,399*	4,650	392	77	280	10	5
2010	67,371	6,060	1,447	237	970	10	5
2011	41,355	1,975	1,428	195	690	10	5
2012	26,746	1,800	1,095	170	690	0	0
2013	19,712	1,460	1,075	150	390	0	0
Total	199,583	15,945	5,437	829	3,020	30	15

Permit numbers: 0642-1536, 0662-1661, 1049-1718, 1071-1770, 1120-1898, 473-1700, 532-1822, 545-1761, 587-1767, 716-1705, 731-1774,753-1599, 774-1714, 781-1824, 782-1719,945-1776, and 965-1821.

Table 14. Humpback whale takes in the North Atlantic.

Year	Approach	Biopsy	Suction cup tagging	Implantable tagging	Acoustic playback
2009	5,260	415	173	45	624
2010	5,568	415	173	45	600
2011	5,468	415	173	45	600
2012	3,614	215	173	45	600
2013	2,770	215	65	0	600
Total	22,680	1,675	757	180	3,024

Permit numbers:10014, 1036-1744, 1058-1733, 1121-1900, 1128-1922, 605-1904, 633-1778, 775-1875, 948-1692, 981-1707, 14586, and 14451.

Table 15. Sperm whale takes in the North Pacific.

Year	Approach	Biopsy	Suction cup tagging	Implantable tagging	Acoustic playback
2009	17,895	770	100	40	0
2010	22,001	1,425	405	170	120
2011	6,300	785	140	345	120
2012	4,416	665	130	325	120
2013	4,416	665	130	325	120
Total	55,028	4,310	905	1,375	480

Permit numbers: 14097, 14122, 14296, 14451, 14534, 727-1915, 14585, 1127-1921, 1071-1770, 473-1700, 540-1811, 731-1774, 781-1824, 782-1719, 0642-1536, 1049-1718, and 774-1714.

Table 16. Cook Inlet beluga whale takes in the North Pacific.

Year	Approach	Biopsy	Tagging	Acoustic playback	Ultrasound
2009	585	0	0	0	0
2010	585	0	0	0	0
2011	0	0	0	0	0
2012	0	0	0	0	0
2013	0	0	0	0	0
Total	1,170	0	0	0	0

Permit numbers: 782-1719.

Table 17. Southern resident killer whale takes in the North Pacific.

Year	Approach	Biopsy	Suction cup tagging	Exhalation sampling
2009	3,050	45	45	105
2010	3,214	55	45	105
2011	2,174	35	45	105
2012	284	10	0	0
2013	284	10	0	0
Total	9,006	155	135	305

Permit numbers: 10045, 14097, 532-1822, 540-1811, 731-1774, 774-1714, 781-1824, 782-1719, 15483, 965-1821, 15483, and 13430.

Table 18. Steller sea lion (eastern DPS) takes in the North Pacific.

Year	Approach	Capture/ restraint	Biopsy	External tag	Mortality	Anaesthesia /drug	Hot brand	Lavage	Ultrasound	Blood/ tissue sample	Tooth extraction
2009	306,001	1,446	980	300	183	1,446	1,350	360	370	840	30
2010	268,752	1,446	980	300	180	1,446	1,350	360	370	840	30
2011	235,752	1,446	980	300	180	1,446	1,350	360	370	840	30
2012	234,952	1,446	980	300	180	1,446	1,350	360	370	840	30
2013	234,952	1,446	980	300	180	1,446	1,350	360	370	840	30
Total	1,280,409	7,230	4,900	1,500	900	7,230	6,750	1,800	1,850	4,200	150

Permit numbers: 13430, 14097, 14325, 14326, 14336, 14337, 15483, 532-1822, 540-1811, 715-1885, 774-1714, and 965-1821.

Table 19. Steller sea lion (western DPS) takes in the North Pacific.

Year	Approach	Capture/ restraint	Biopsy	External tag	Mortality	Anaesthesia /drug	Hot brand	Lavage	Ultrasound	Blood/ tissue sample	Tooth extraction
2009	205,638	4,698	1,032	232	25	782	570	265	2,322	778	20
2010	154,819	4,698	1,032	232	16	782	570	265	2,322	778	20
2011	154,819	4,698	1,032	232	16	782	570	265	2,322	778	20
2012	154,769	4,698	1,032	232	16	782	570	265	2,322	778	20
2013	154,769	4,698	1,032	232	16	782	570	265	2,322	778	20
Total	824,814	23,490	5,160	1,160	89	3,910	2,850	1,325	11,610	3,890	100

Permit numbers: 1118-1881, 1119-1882, 14296, 14324, 14325, 14326, 14327, 14329, 14335, 14336, 14337, 715-1885, and 965-1821.

Effects of the Proposed Action

Pursuant to section 7(a)(2) of the ESA, federal agencies must ensure, through consultation with the NMFS, that their activities are not likely to jeopardize the continued existence of any listed species or result in the destruction or adverse modification of critical habitat. The proposed issuance of permit 14245 would authorize "takes" by harassment of marine mammals during the proposed research by the NMML by directed and unintentional approach, satellite and sensory tagging, biopsy, and photoidentification. In this section, we describe the potential physical, chemical, or biotic stressors associated with the proposed actions, the probability of individuals of listed species being exposed to these stressors based on the best scientific and commercial evidence available, and the probable responses of those individuals (given probable exposures) based on the available evidence. As described in the *Approach to the Assessment* section, for any responses that would be expected to reduce an individual's fitness (i.e., growth, survival, annual reproductive success, or lifetime reproductive success), the assessment would consider the risk posed to the viability of the population(s) those individuals comprise and to the listed species those populations represent. The purpose of this assessment and, ultimately, of this Opinion is to determine if it is reasonable to expect the proposed action to have effects on listed species that could appreciably reduce their likelihood of surviving and recovering in the wild.

For this consultation, we are particularly concerned about behavioral and physiological disruptions that may result in animals that fail to feed or breed successfully or fail to complete their life history because these responses are likely to have population-level consequences. The ESA does not define harassment nor has the NMFS defined the term pursuant to the ESA through regulation. However, the Marine Mammal Protection Act of 1972, as amended, defines harassment as any act of pursuit, torment, or annoyance which has the potential to injure a marine mammal or marine mammal population in the wild or has the potential to disturb a marine mammal or marine mammal population in the wild by causing disruption of behavioral patterns, including, but not limited to, migration, breathing, nursing, breeding, feeding, or sheltering [16 U.S.C. 1362(18)(A)]. The latter portion of this definition (that is, "...causing disruption of behavioral patterns including...migration, breathing, nursing, breeding, feeding, or sheltering") is almost identical to the U.S. Fish and Wildlife Service's regulatory definition of "harass"[2] pursuant to the ESA. For this Opinion, we define harassment similarly: an intentional or unintentional human act or omission that creates the probability of injury to an individual animal by disrupting one or more behavioral patterns that are essential to the animal's life history or its contribution to the population the animal represents.

Our analysis considers that behavioral harassment or disturbance is not limited to the "take" definition and may in fact occur in many ways. Fundamentally, if our analysis leads us to conclude that an individual changes its behavioral state (for example, from resting to traveling away from the approaching vessel or from traveling to evading), we consider the individual to have been harassed or disturbed, regardless of it has been approached closely enough to breach recommended stand-off boundaries established under authority of the MMPA. In addition, individuals may respond in a variety of ways, some of which have more significant fitness

2 An intentional or negligent act or omission which creates the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering (50 CFR 17.3)

consequences than others. For example, evasion of an approaching vessel would be more significant than slow travel away from the same stressor due to increased metabolic demands, stress responses, and potential for habitat abandonment that this response could or would entail. As described in the *Approach to the assessment*, the universe of likely responses is considered in evaluating the fitness consequences to the individual and (if appropriate), the affected population and species as a whole to determine the likelihood of jeopardy.

Potential stressors

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

1. aerial transit during proposed activities
2. surface vessel transit during proposed activities
3. close approaches to listed whales and species proposed for listing by research vessels
4. application of telemetry tags
5. continued attachment of tags
6. collection of blubber biopsy samples

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

The NMML proposes to conduct aerial surveys, vessel surveys, biopsy sampling, and tagging of cetaceans in waters of the Arctic, North Atlantic, and North Pacific oceans that would target numerous cetacean species, including listed blue, fin, sei, humpback, bowhead, North Pacific right, Cook Inlet beluga, southern resident killer, and sperm whales, as well as the proposed Hawaiian insular population of false killer whales. Although proposed for direct research, individuals of these species could incidentally be exposed to stressors associated with the proposed action, such as the potential for shipstrike and acoustic noise exposure. Operators and observers will search for marine mammals while underway and we feel confident in the ability of operators to locate, identify, and avoid direct contact with individuals. While in close proximity to marine mammals in undertaking the proposed research, operators would be moving slowly and deliberately in ways in which the vessels would approach, but not physically contact listed or proposed marine mammals. We do not expect that vessel transits pose a significant risk of shipstrike to listed proposed marine mammals under the proposed actions for reasons. We therefore discount the potential for shipstrike in association with the proposed actions to target species.

The research vessels would produce noise in the acoustic environment which has the potential to mask the vocalizations produced by these species or other significant acoustic information, introducing the possibility that important sounds may not be perceived by individuals near the research vessel (particularly when operating at high speed). However, researchers would be visually searching for cetaceans and avoiding close approaches of all but target individuals. Exposure to masking sounds is expected to be brief and discountable to listed cetaceans.

Aerial surveys proposed by the applicant would co-occur with haul-out areas of some listed or proposed for listing pinnipeds, including both eastern and western DPSs of Steller sea lion, the Beringia DPS of bearded seals, and Arctic as well as Okhotsk DPSs of ringed seal, as well as co-occur in time and space with North Atlantic right whales. These species may respond to the sight or sound of the aircraft, causing them to at least temporarily abandon their haul-out or change their behavioral state. Therefore, unintentional harassment of these listed or proposed for listing species is possible.

Accordingly, this consultation focused on the following stressors likely to occur from the proposed seismic activities and may adversely affect ESA-listed species: Overflights of marine mammals by survey aircraft; close approaches to whales by research vessels; application and continued attachment of tags; continued attachment of tags; and collection of blubber biopsy samples.

Exposure analysis

The proposed action is to permit biopsy, suction-cup tag, satellite tag, close approach, photo-ID, and behavioral observations of marine mammals as outlined in Table 20.

Table 20. Number of animals proposed to be taken by species, life stage, and action under Permit 14245.

Species-population	Life stage	Proposed Number of individuals taken annually	Action
Blue whale (*Balaenoptera musculus*)-North Pacific	All	4,000	Close approach-aerial
		2,010	Close approach-vessel
	Juvenile/adult	1,600	Close approach-vessel, biopsy, suction cup tagging
		150	Close approach-vessel, biopsy, any style tagging
	Non-neonate calf <6 months old primarily, >6 months secondarily	240	Close approach-vessel, biopsy, suction cup tagging
Fin whale (*Balaenoptera physalus*)- North Pacific	All	8,000	Close approach-aerial
		4,250	Close approach-vessel
	Juvenile/adult	2,000	Close approach-vessel, biopsy, suction cup tagging
		300	Close approach-vessel, biopsy, any style tagging
	Non-neonate calf <6 months old primarily, >6 months secondarily	450	Close approach-vessel, biopsy, suction cup tagging
Sei whale	All	1,000	Close approach-aerial

(*Balaenoptera borealis*)-North Pacific		682	Close approach-vessel
	Juvenile/adult	300	Close approach-vessel, biopsy, suction cup tagging
		10	Close approach-vessel, biopsy, any style tagging
	Non-neonate calf <6 months old primarily, >6 months secondarily	8	Close approach-vessel, biopsy, suction cup tagging
Humpback whale (*Megaptera novaeangliae*)-North Pacific	All	8,000	Close approach-aerial
		3,645	Close approach-vessel
	Juvenile/adult	3,000	Close approach-vessel, biopsy, suction cup tagging
		380	Close approach-vessel, biopsy, any style tagging
	Non-neonate calf <6 months old primarily, >6 months secondarily	525	Close approach-vessel, biopsy, suction cup tagging
Humpback whale (*Megaptera novaeangliae*)-North Atlantic	All	1,000	Close approach-aerial
		375	Close approach-vessel
	Juvenile/ adult	500	Close approach-vessel, biopsy, any style tagging
		50	Close approach-vessel, biopsy, suction cup tagging
	Non-neonate calf <6 months old primarily, >6 months secondarily	75	Close approach-vessel, biopsy
Bowhead whale (*Balaena mysticetus*)-Arctic	All	11,000	Close approach-aerial
		650	Close approach-vessel
	Juvenile/ adult	1,000	Close approach-vessel, biopsy, suction cup tagging
		200	Close approach-vessel, biopsy, any style tagging
	Non-neonate calf <6 months old primarily, >6 months secondarily	150	Close approach-vessel, biopsy, suction cup tagging
North Pacific right whale (*Eubalaena glacialis*)-North Pacific	All	200	Close approach-aerial
		130	Close approach-vessel
	Juvenile/ adult	50	Close approach-vessel, biopsy, suction cup tagging
		10	Close approach-vessel, biopsy, any style tagging

	Non-neonate calf <6 months old primarily, >6 months secondarily	10	Close approach-vessel, biopsy, suction cup tagging
Sperm whale (*Physeter macrocephalus*)- North Pacific	All	8,000	Close approach-aerial
		2,500	Close approach-vessel
	Juvenile/ adult	2,000	Close approach-vessel, biopsy, suction cup tagging
		200	Close approach-vessel, biopsy, any style tagging
	Non-neonate calf <6 months old primarily, >6 months secondarily	300	Close approach-vessel, biopsy, suction cup tagging
Cook Inlet beluga whale (*Delphinapterus leucas*)- Cook Inlet	All	11,700	Close approach-aerial
		670	Close approach-vessel
	Juvenile/ adult	300	Close approach-vessel, biopsy, suction cup tagging
False killer whale (*Pseudorca crassidens*)- Hawaiian insular	All	100	Close approach-aerial
		90	Close approach-vessel
	Older than neonate	100	Close approach-vessel, biopsy, suction cup tagging
	Juvenile/ adult	10	Close approach-vessel, biopsy, any style tagging
Southern resident killer whale (*Orcinus orca*)- Southern resident	All	500	Close approach-aerial
		490	Close approach-vessel
	Juvenile/ adult	10	Close approach-vessel, biopsy, suction cup tagging
Steller sea lion (*Eumetopias jubatus*)-Eastern DPS	All	10,000	Incidental disturbance via close approach-aerial
Steller sea lion (*Eumetopias jubatus*)-Western DPS	All	10,000	Incidental disturbance via close approach-aerial
Bearded seal (*Erignathus barbatus nauticus*)- Beringia DPS	All	1,000	Incidental disturbance via close approach-aerial

Ringed seal (*Phoca hispida hispida*)-Arctic DPS	All	2,000	Incidental disturbance via close approach-aerial

The NMML expects that an individual of any age class or sex may be approached by vessel up to 15 times per year under the proposed activities. No neonates would be allowed to be biopsied, but juveniles, subadult, and adult age classes of any sex may be tagged with up to two tags that break the skin and one additional non-penetrating tag simultaneously. The NMML expects that aerial surveys may conceivable overfly an individual 126 times per year (example: Cook Inlet beluga whales). The permit would be conditioned to minimize harassment from tagging and biopsy activities to no more than three times per day. However, the applicant expects that tagging and biopsy would be successful within two attempts for each activity; monitoring reports indicate misses for both activities are unusual to rare. Large whale species would not carry more than three tags in a given year; smaller cetaceans would not be tagged more than once. Permit conditions also stipulate that aerial surveys must be flown at altitudes above 229 m, but may briefly descend to 92 m and that if a response to aerial survey is observed, the survey aircraft must leave the area. All biopsy tips must be disinfected prior to use. Mothers with neonatal calves may not be tagged. Tagging or biopsy attempts must be discontinued if repetitive strong reactions are found. Calves, mothers with calves, and emaciated or otherwise unhealthy-looking individuals will not be targeted, so exposure to these groups is unlikely to occur.

Although these activities are proposed to occur, we expect the level of exposure to these activities for most listed species to be much less than the levels of "take" requested above (Table 20). This is based upon annual monitoring reports of NMML activities from 1998-2010 that include activities similar or identical to those proposed permit. Expected exposure levels for each species and activity were determined by calculating means and standard deviations for each activity to each species. Four standard deviations were added to each mean for which sufficient data were available to encompass a reasonably likely maximum exposure to similar activities for each species in the future (Table 21). In addition, we assume 4% annual population growth for all populations that are currently increasing in abundance. We provisionally accept the levels of "take" as the level of exposure for species for which we lack data from NMML monitoring reports. We expect these values, as well as the values estimated for previous activities, will be adjusted in the future based upon future NMML annual reports.

Table 21. Expected annual exposure levels for listed species to proposed activities.

Species-population	**Expected annual exposure**	**Basis (AR= NMML annual reports; PA= provisionally accept "take" estimates; FR= field reports)**	**Activity**
Blue whale (*Balaenoptera musculus*)-North Pacific	0	AR	Close approach-aerial, vessel
	0	AR	Biopsy
	0	AR	Tagging

Fin whale (*Balaenoptera physalus*)- North Pacific	294	AR	Close approach-aerial
	152	AR	Close approach-vessel
	34	AR	Biopsy
	4	AR	Tagging
Sei whale (*Balaenoptera borealis*)-North Pacific	0	AR	Close approach-aerial
	0	AR	Close approach-vessel
	0	AR	Biopsy
	0	AR	Tagging
Humpback whale (*Megaptera novaeangliae*)- North Pacific	240	AR	Close approach-aerial
	9,537	AR	Close approach-vessel
	2,215	AR	Biopsy
	26	AR	Tagging
Humpback whale (*Megaptera novaeangliae*)- North Atlantic	1,000	PA	Close approach-aerial
	375	PA	Close approach-vessel
	75	PA	Close approach-vessel, biopsy
	500	PA	Close approach-vessel, biopsy, any style tagging
	50	PA	Close approach-vessel, biopsy, suction cup tagging
Bowhead whale (*Balaena mysticetus*)-Arctic	2,198	AR	Close approach-aerial
	34	AR	Close approach-vessel
	29	AR	Biopsy
	36	AR	Tagging
North Pacific right whale (*Eubalaena*	15	AR	Close approach-aerial
	8	AR	Close approach-vessel

glacialis)-North Pacific	9	AR	Biopsy
	8	AR	Tagging
Sperm whale (*Physeter macrocephalus*)-North Pacific	0	AR	Close approach-aerial
	106	AR	Close approach-vessel
	35	AR	Biopsy
	12	AR	Tagging
Cook Inlet beluga whale (*Delphinapterus leucas*)- Cook Inlet	6,641	FR	Close approach-aerial
	34	AR	Close approach-vessel
	23	AR	Close approach-vessel, biopsy
	30	AR	Close approach-vessel, tagging
False killer whale (*Pseudorca crassidens*)-Hawaiian insular	100	PA	Close approach-aerial
	90	PA	Close approach-vessel
	100	PA	Close approach-vessel, biopsy, suction cup tagging
	10	PA	Close approach-vessel, biopsy, any tagging
Southern resident killer whale (*Orcinus orca*)-Southern resident	500	PA	Close approach-aerial
	500	PA	Close approach-vessel
	10	PA	Close approach-vessel, biopsy, suction cup tagging
Steller sea lion (*Eumetopias jubatus*)-Eastern DPS	249	AR	Unintentional harassment
Steller sea lion (*Eumetopias jubatus*)-Western DPS	0	AR	Unintentional harassment

Bearded seal (*Erignathus barbatus nauticus*)-Beringia DPS	0	AR	Unintentional harassment
Ringed seal (*Phoca hispida hispida*)-Arctic DPS	0	AR	Unintentional harassment

Although the NMML is proposing significant levels of exposure to aerial/vessel approach, biopsy, and/or tagging activities for sei whales, western DPS Steller sea lions, Beringia bearded seals, and Arctic ringed seals, we do not expect that these species will be exposed to these activities under the proposed permit. Although the NMML has engaged in extensive activities within the action area since 1998 similar to those currently proposed, the aforementioned species have not been documented to be sighted by the NMML during these activities (even less approached, biopsied, or tagged, which must be reported within annual reports per permit conditions). Similarly, sperm whales have not been observed during aerial surveys and we do not expect their exposure here. Blue whales have not been "taken," and as far as we can tell exposed to approaches by the NMML for the last 12 years, but recent evidence regarding blue whale distribution supports the possibility that blue whales may co-occur with proposed activities with greater frequency (Calambokidis et al. 2009). Therefore, we expect one or a few blue whales might be exposed to approaches during the life of the permit.

Although the NMML estimated the number of exposures to individuals, the NMML cannot determine one individual from another in the field. Therefore, we used a Poisson distribution to estimate the number of exposures a given individual would experience annually as well as over the life of the proposed permit based upon the levels of expected exposure (Table 22). We do not have data to determine the likely exposure of populations not previously or extensively exposed to prior NMML research activities, such as Hawaiian insular false killer whales, southern resident killer whales, and humpback whales in the North Atlantic; for these populations, we provisionally accept the NMMLs estimates of individual exposure.

Table 22. Expected exposures of proposed activities to individuals annually and over the permit's duration.

Species/activity	**Annual**		**Over permit's lifespan**	
	Expected individual exposure	**Range of individual exposures**	**Expected individual exposure**	**Range of individual exposures**
Blue whale (*Balaenoptera musculus*)-North Pacific				
Aerial approach	0	n/a	0	n/a
Vessel approach	0	n/a	0	n/a
Tagging	0	n/a	0	n/a
Biopsy	0	n/a	0	n/a
Fin whale (*Balaenoptera physalus*)- North Pacific				

Aerial approach	1	0-2	1	0-2
Vessel approach	1	0-2	1	0-2
Tagging	1	0-1	1	0-1
Biopsy	1	0-1	1	0-2
Sei whale (*Balaenoptera borealis*)-North Pacific				
Aerial approach	0	n/a	0	n/a
Vessel approach	0	n/a	0	n/a
Tagging	0	n/a	0	n/a
Biopsy	0	n/a	0	n/a
Humpback whale (*Megaptera novaeangliae*)-North Pacific				
Aerial approach	1	0-2	1	0-2
Vessel approach	1	0-4	1	0-6
Tagging	1	0-1	1	0-1
Biopsy	1	0-2	1	0-3
Bowhead whale (*Balaena mysticetus*)-Arctic				
Aerial approach	1	0-2	1	0-3
Vessel approach	1	0-1	1	0-2
Tagging	1	0-1	1	0-2
Biopsy	1	0-1	1	0-2
North Pacific right whale (*Eubalaena glacialis*)-North Pacific				
Aerial approach	1	0-2	1	0-3
Vessel approach	1	0-2	1	0-2
Tagging	1	0-2	1	0-2
Biopsy	1	0-2	1	0-3
Sperm whale (*Physeter macrocephalus*)-North Pacific				
Aerial approach	0	n/a	0	n/a
Vessel approach	1	0-2	1	0-2
Tagging	1	0-1	1	0-1
Biopsy	1	0-1	1	0-2
Cook Inlet beluga whale (*Delphinapterus leucas*)- Cook Inlet				
Aerial approach*	9	0-19	41	0-95
Vessel approach	1	0-2	1	0-3
Tagging	1	0-2	1	0-3
Biopsy	1	0-2	1	0-3
Steller sea lion (*Eumetopias jubatus*)-Eastern DPS				
Aerial approach	0	0-1	1	0-2
Steller sea lion (*Eumetopias jubatus*)-Western DPS				
Aerial approach	0	n/a	0	n/a
Bearded seal (*Erignathus barbatus nauticus*)-Beringia DPS				
Aerial approach	0	n/a	0	n/a

Ringed seal (*Phoca hispida hispida*)-Arctic DPS				
Aerial approach	0	n/a	0	n/a

*Due to the large sample size and relatively high mean-to-variance ratio of these data, the Poisson distribution was not applied to aerial survey approaches of Cook Inlet beluga whales. Instead, a normal distribution was calculated.

Response analysis

As discussed in the *Approach to the assessment* section of this Opinion, response analyses determine how listed resources are likely to respond after exposure to an action's effects on the environment or directly on species themselves. For the purposes of consultation, our assessments try to detect potential lethal, sub-lethal (physiological), or behavioral responses that might result in reducing the fitness of listed individuals. Ideally, response analyses would consider and weigh evidence of adverse consequences as well as evidence suggesting the absence of such consequences.

There is mounting evidence that wild animals respond to human disturbance in the same way that they respond to predators (Beale and Monaghan 2004; Frid 2003; Frid and Dill 2002; Gill et al. 2001; Harrington and Veitch 1992; Lima 1998; Romero 2004). These responses manifest themselves as stress responses (in which an animal perceives human activity as a potential threat and undergoes physiological changes to prepare for a flight or fight response or more serious physiological changes with chronic exposure to stressors), interruptions of essential behavioral or physiological events, alteration of an animal's time budget, or some combinations of these responses (Frid and Dill 2002; Romero 2004; Sapolsky et al. 2000; Walker et al. 2005). These responses have been associated with abandonment of sites (Sutherland and Crockford 1993), reduced reproductive success (Giese 1996; Mullner et al. 2004), and the death of individual animals (Bearzi 2000; Daan 1996; Feare 1976). Stress is an adaptive response and does not normally place an animal at risk. However, distress involves a stress response resulting in a biological consequence to the individual. The mammalian stress response involves the hypothalamic-pituitary-adrenal (HPA) axis being stimulated by a stressor, causing a cascade of physiological responses, such as the release of the stress hormones adrenaline (epinephrine), glucocorticosteroids, and others (Busch and Hayward 2009)(Gulland et al. 1999; Morton et al. 1995; St. Aubin and Geraci 1988; St. Aubin et al. 1996; Thomson and Geraci 1986). These hormones subsequently can cause short-term weight loss, the liberation of glucose into the blood stream, impairment of the immune and nervous systems, elevated heart rate, body temperature, blood pressure, and alertness, and other responses (Busch and Hayward 2009; NMFS 2006g)(Cattet et al. 2003; Delehanty and Boonstra 2009; Elftman et al. 2007; Fonfara et al. 2007; Kaufman and Kaufman 1994; Mancia et al. 2008; Moe and Bakken 1997; Noda et al. 2007; Thomson and Geraci 1986)(Dierauf and Gulland 2001; Omsjoe et al. 2009a). In some species, stress can also increase an individual's susceptibility to gastrointestinal parasitism (Greer et al. 2008). In highly-stressful circumstances, or in species prone to strong "fight-or-flight" responses, more extreme consequences can result, including muscle damage and death (Cowan and Curry 1998b; Cowan and Curry 2002; Cowan and Curry 2008; Herraez et al. 2007). The most widely-recognized indicator of vertebrate stress, cortisol, normally takes hours to days to return to baseline levels following a significantly stressful event, but other hormones of the HPA axis may persist for weeks (Dierauf and Gulland 2001). Mammalian stress levels can vary by age, sex, season, and health status (Gardiner and Hall 1997; Hunt et al. 2006; Keay et al. 2006; Kenagy and Place 2000; Nunes et al. 2006; Romero et al. 2008; St. Aubin et al. 1996). Smaller mammals tend to react more strongly to stress than larger mammals (Peters 1983); a trend

reflected in data from Gauthier and Sears (1999) where smaller whale species tended to react more frequently to biopsy than larger whales. Stress is lower in immature right whales than adults and mammals with poor diets or undergoing dietary change tend to have higher fecal cortisol levels (Hunt et al. 2006; Keay et al. 2006; Kitaysky and Springer 2004).

Close approach-aerial surveys

Few published data are available to evaluate the responses of listed marine mammals to aircraft overflights. Malme et al. (1983a) made an opportunistic evaluation on a bowhead whale group. In this event, a circling single-engine aircraft descended from roughly 400 m (above the normal altitude generally used in proposed aerial surveys) to 60 m (well below the minimum altitude proposed for permitted aerial surveys). Once the aircraft descended and approached the whales at its closest point, the group discontinued its behavior and split into two groups. The groups rejoined and continued their prior behavior immediately after the departure of the aircraft. Richardson et al. (1985a) found bowheads to respond frequently to Islander survey aircraft approaches below 305 m, infrequently at 457 m, and not at all at 610 m; responses were normally hasty dives and sometimes gradual departure from the area. Blow interval may also decrease upon aircraft descent. He also cites Marquette et al. (1982) as bowheads rarely reacting in a negative manner to aircraft flying as low as 75 m. Richardson et al. (1985a) further cites Ljungblad et al. (1980) and Ljungblad (1981) as bowhead responses being variable by date and whale activity, with mating whales being less responsive than when they were not. Payne et al. (1983) found southern right whales to rarely react strongly to survey aircraft flying at 65-130 m. Richter et al. (2006) found sperm whales (specifically transient sperm whales) to briefly increase their time at the surface and take 20 seconds longer during their dives to start “clicking” (presumably related to prey detection), although they determined that their findings were not biologically meaningful. They did note that habituation to both vessel and aerial approaches likely occurred in “resident” individuals. Luksenburg and Parsons (2009) found that across cetacean species, most respond (when they respond) by diving. Smaller groups respond strongly less often than do larger ones; individuals in shallow water respond more frequently than those in deep water, as do mothers with calves versus other group types, when individuals were initially resting or milling, and when aircraft fly at lower altitude. Sperm whales responded in 28% (7 of 25) cases to survey aircraft (mostly by diving) and false killer whales responded in <29% of overflights (Smultea et al. 2008). Overflight and circling at 235-335 m above a sperm whale group by a Skymaster survey aircraft elicited appears to have elicited a group defensive formation from a sperm whale pod. Bowhead whales responded in 2.2% of 507 observations to Twin Otter overflights, with most responses of short surfacing, abrupt dives, or heading away from the plane when the aircraft was flying at below 182 m and less than 250 m laterally from target individuals (Patenaude et al. 2002). Beluga whales have been found to respond in 3.2% of 760 overflights by immediately diving with a tail thrash, unusual turns or changes in heading, turning to look upwards, or other behavioral reactions. Most responses were from the same aircraft type and at the same ranges previously mentioned for bowheads, with direct overflights causing the most conspicuous responses. After measuring sound detected via hydrophone during aircraft overflights, it was determined that bowheads would likely hear a plane flying directly overhead at 150 or 300 m altitude, but belugas could perhaps just barely hear a plane at 300 m. An aircraft’s shadow may cause cetaceans to respond as well (Luksenburg and Parsons 2009).

Fewer data are available regarding pinnipeds responses. Southwell (2005) found that alert and

movements by seals increased once the survey aircraft was closer than 800 m away. However, this was from a helicopter, which has much different speed and sound characteristics than does a survey plane. The NMML has found that pinnipeds may be disturbed by entering the water when overflown, although this has been limited to two Steller sea lions. These species may also or alternatively lift their heads to observe the aircraft. The NMML has found that <10% of Steller sea lions, <20 of ringed seals, and <10% of bearded seals respond to aircraft overflights, with individuals in water responding at an even lower frequency.

The NMML has been conducting aerial surveys similar or identical to those proposed under the present permit for over a decade. As a result, the NMML has documented the numbers of individuals responding to overflights as part of their required documentation in annual reports. Analysis of response data from 1998-2010 for target species indicates that apparent response is rare. Based upon methods previously described to calculate maximum likely exposure and the paucity of documented response from available literature and expert opinion (Laura Morse, NMFS, pers. comm.; Trisia Naessig, pers. comm.), we do not anticipate fin, sei, humpback, North Pacific right, or sperm whales to respond to aerial surveys during the course of a year. We also do not expect Arctic DPS ringed seals or western DPS Steller sea lions to respond to aircraft overflights. We expect up to six bowhead and 25 Cook Inlet beluga whales to respond in ways documented in the literature, namely startle responses, rapid dives, changes in direction. One or a few blue whales might respond in a similar manner to overflights as well based upon recently documented distributional shifts (Calambokidis et al. 2009). Although data for Hawaiian insular false killer whales and southern resident killer whales is generally lacking, a few individuals may also respond to aircraft overflights in a similar manner. We expect up to 25 eastern DPS Steller sea lions may respond to overflights with head lifts or entering the water. Additional responses based upon additional exposures (identified in Table 22 on pages 113-114) are also possible; however, assuming individuals experience re-exposure, we expect responses necessarily will not occur with every re-exposure and will vary by individual and context.

Close approaches-surface vessel

Vessel approaches have the potential to induce behavioral and physiological changes in individuals being targeted. The degree to which individuals are disturbed is highly variable. Whales may respond differently depending upon what behavior the individual or pod is engaged in before the vessel approaches (Hooker et al. 2001c; Wursig et al. 1998), the degree to which they have become accustomed to vessel traffic (Richter et al. 2006), and between species or individuals (Gauthier and Sears 1999). Overall, reactions include little to no observable change in behavior to momentary changes in swimming speed, pattern, orientation, diving and time spent submerged, foraging, respiratory patterns, and may include aerial displays like breaching and lobtailing (Baker and Herman. 1989; Best et al. 2005; Brown et al. 1991b; Clapham and Mattila 1993; Jahoda et al. 2003). Jahoda et al. (2003) found effects of more than a few minutes, with fin whales failing to return to baseline behaviors after one hour of observation in some cases, in spite of the fact that Gauthier and Sears (1999) found fin whales to be less responsive than humpbacks.

North Atlantic right whales (taxonomically similar to North Pacific right and bowhead whales) may not respond at all to kayaks, sailing sloops, or steel-hulled diesel-powered vessels approaching within five meters, although other individuals (possibly under different contexts)

have responded to the same diesel-powered vessel from 50 m away, usually by turning away from the path of the ship (Goodyear 1993a). Baumgartner and Mate (2003b) found that 71% of 42 North Atlantic right whales approached (and sometimes tagged) in a rigged inflatable boat within 10 m did not overtly respond. Of those that did respond, behaviors included head lifts and lunges, back arching, rolling, and fluke beats. Feeding dive durations were also shorter by 13-17% in the dive following approach/tagging, but no difference was found in the duration of subsequent dives. Mate et al. (1997a) found that although North Atlantic right whales generally responded to and avoided close approach, the level of response varied. Watkins (1986) found that whales are more responsive to approach when they are inactive and less responsive when feeding or socializing.

Humpback whales have been the best-studied whale species in regards to responses to close approaches by vessels. Numerous studies have documented varied responses of humpback whales to vessel approaches, ranging from no response to approach to evasion (Goodyear 1993a; Salden 1993). In response to vessel approach, Felix et al. (2001) found that 27 of 86 individuals approached resulted in avoidance of the vessel (50 were indifferent and 9 approached vessels), including long dive, change in heading, tail splashes, altered swimming speed or breathing frequency, and group structure disruption. Approaching vessels may instigate aerial behavior, such as fluke slapping and breaching, behavior recently suggested to be a switch in communication from vocal to surface active signaling (Baker et al. 1983a; Baker et al. 1983c; Baker et al. 1982; Dunlop et al. 2009; Holt et al. 2009). Hall (1982) did not find social or feeding behavior to be disturbed by vessel traffic or close approaches. However, there is the possibility that humpback whales may habituate to vessel noise if given sufficient time and exposure (Clapham and Mattila 1993; Watkins 1986). Goodyear (1993a) did not observe changes in behavior due to vessel approaches in most cases, although an increase in speed did occur on one occasion when a whale was approached within 10 m. Cantor et al. (2010) generally found resting or socializing whales to switch to traveling upon approach of their research vessels. Watkins et al. (1981) found that humpback whales appeared to react to vessel approach by increasing swim speed, exhibiting a startle reaction, and moving away from the vessel with strong fluke motions. Baker and Herman (1989), Baker et al. (1982) and (1983a; 1983c), Bauer (1986), Bauer and Herman (1986), and Green and Green (1990) found that humpbacks spent less time at the surface and altered their direction of travel in response to approaching vessels. Increased time underwater and decreased swim speed persisted for up to 20 minutes after vessels left the area. Watkins and Goebel (1984) found humpbacks to be very difficult to approach, possibly due to physical ocean features in the area that likely altered sound properties such that vessel noise was difficult to detect except at close range, resulting in whales suddenly becoming aware of boats in close proximity and reacting strongly as a result. Norris (1994) documented changes in humpback song structure in response to passing vessels, with unit and phrase durations reduced versus control periods.

Bauer and Herman (1986) studied the potential consequences of vessel disturbance on humpback whales wintering off Hawaii. They as well as Scheidat et al. (2004) and Hemphill et al. (2006) noted changes in respiration, diving, swimming speed (50-300%) and direction, social exchanges, and other behavioral changes correlated with the number, speed, direction, and proximity of vessels. Agonistic behavior has also been noted (Bauer and Herman 1986). Results of vessel approach were different depending on individual sex and age class (smaller groups and groups with calves appeared more responsive), but humpback whales generally tried to avoid

vessels beginning at 500 to 1,000 m away. Similar results were found in Alaskan waters, with increased dive durations and orientation away from the path of moving boats, often at ranges up to 3-4 km (Baker et al. 1983b; Baker and Herman. 1989). Approaches in Alaskan waters closer than 100 m initiated evasive behavior (Hall 1982); Watkins (1986) found little response to approaches outside of 100 m away, although humpbacks regularly reacted to outboard vessels on a collision course even from long distance.

Responses can also change over long timeframes; Watkins (1986) looked at whale responses off Cape Cod over a several decade period and found that humpbacks shifted their general response from being generally evasive to a tendency to approach vessels. Mizroch et al. (2010) followed-up on several humpback whales that were approached and radio tagged over the course of several decades. They found no basis for substantiating a long-term reaction to approach, including gross measures of growth and reproduction.

Information on contextual responses is also relatively abundant for humpback whales. Responses by humpback whales likely depend upon a given individual's prior experience and current situation (Clapham and Mattila 1993). The use of smaller, outboard-powered vessels (presumably louder) elicited more frequent and stronger responses to biopsy attempts than larger, inboard-powered vessels; sex was not a factor in response frequency or intensity (Cantor et al. 2010). Sudden changes in vessel speed and direction have been identified as contributors to humpback whale behavioral responses from vessel maneuvering (Watkins 1981b). The more active the group, the more easily it was disturbed; however, Cantor et al. (2010) found structuring in the response rate of various individuals in mating groups, with male response becoming progressively less frequent with increasing degree of dominance in the mating group. Mother-calf pairs were the most easily disturbed group, followed by all adult groups, adult-subadult mixes, and all subadult groups (Felix 2001). Weinrich et al. (1991) and (1992b), Cantor et al. (2010), as well as Krieger and Wing (1984) found feeding animals to be least responsive, although data from these studies was contradictory when evaluating responses while resting or on breeding grounds. The Weinrich studies also found that respiratory parameters are not good indicators of responsiveness due to the large natural variance associated with them. However, numerous studies have identified significant changes in respiration and diving in association with vessel traffic (see Bauer and Herman (1986) for a summary). On several occasions, research trips conducted by Krieger and Wing (1984) had to actively avoid collisions with humpbacks, although whales presumably were aware of the vessel's presence. Single or paired individuals may respond more than larger groups (Bauer and Herman 1986). Würsig et al. (1998) found milling or resting cetaceans to be more sensitive.

Repeated exposure can have a cumulative effect that is greater than the sum of individual exposures, eliciting responses that are more significant for individuals and populations, although Cantor et al. (2010) did not find a difference in response based upon re-exposure. However, humpback whales have vacated areas where relatively high boat traffic and human activity occurs (Herman 1979). Major declines and distributional shifts in Glacier Bay, Alaska were correlated with a rapid and significant increase in vessel traffic from 1976 to 1978, whereas humpback whales in other nearby areas with less traffic did not undergo such changes (Bauer and Herman 1986). It should be noted that potentially reduced prey resources may also have been important in this redistribution (Bauer and Herman 1986). Matkin and Matkin (1981) did not find a correlation between humpback whale behavior and recreational vessels.

Other large whale species have also been investigated for their responses to close vessel approaches. Bowheads seem to be particularly sensitive, with individuals swimming rapidly away (rarely seen as a natural behavior) and reducing dive and surface cycles in response to a crew boat used to study whales in Arctic waters at ranges of 1-4 km, with individuals moving up to 2-3 km away (Richardson et al. 1985a). Movement away still occurred when engines were disengaged and idling at ranges greater than 900 m, but no effect was found when engines were off. Individuals would also scatter from their groups, a condition that would persist well after the vessel had vacated the area and hamper echelon feeding. Gray whales may be more sensitive to approach while resting; they frequently startle in response to close approach and swim rapidly away (Mate and Harvey 1983). Pettis et al. (1999) found gray whales tended to disperse in the presence of boats and aggregate in their absence. When directly approached, individuals were more likely to change heading, do a fluke-down dive, or slip under water, whereas indirect approaches tended to result in fluke or flipper swishes and head raises. Calf presence did not appear to impact response, although calves tended to respond with bubble release from the blowholes, change their heading, or roll, whereas adults were more likely to dive or slip underwater. Gray whales vacated a wintering (breeding, non-feeding) lagoon apparently in response to increased commercial vessel traffic but reoccupied it after vessel traffic decreased (Reeves 1977). Such impacts can interfere with the reproductive success of individual whales and the populations they represent (Croll et al. 2001b). Fin whales were found to accelerate their speed upon vessel approach (Watkins 1981b). Fin whales were particularly evasive in a study published by Ray et al. (1978), exhibiting high-speed swimming, frequent changes in heading, separation of groups, and irregular breathing patterns. As with humpback whales, fin whales have been found to respond by rapid course change, accelerated dive, and speed increases to vessel noise, particularly throttle changes, such as reversing. Recognition (sensitization) of tagging vessels by both humpback and fin whales has not been seen to occur.

Cook Inlet beluga whales are highly evasive of small boat approaches, likely at least partly due to a history of subsistence hunting from small boats in Cook Inlet. (Ferrero et al. 2000). Approaches to within 10 m instigate evasive behavior, which seems directed towards deeper water and may displace individuals from seasonal foraging locations, although habitat abandonment has not been observed (Ferrero et al. 2000; Lerczak et al. 2000; Shelden 1995). When active approaches were conducted, whales would move 300-500 m away from the research vessel, but return to the area within minutes of discontinuing pursuit (Shelden 1995). Based upon prior research, individuals appear to learn the sound characteristics of at least research vessels if not others and become more evasive sooner in subsequent encounters (Barbara Mahoney, NMFS, pers. comm. in Shelden (1995)). Additionally, surfacing behavior switches from a rolling surfacing behavior to one where only the top of the head is lifted above the water (Shelden 1995). Individuals increase their surface speed for one to two minutes, after which surfacing become more frequent. Eventually, individuals appear to tire and slow their pace, at which time tagging in the past have taken place. Tagging bouts have taken an average of 2.7 minutes in the past, with a maximum of 10 minutes (Shelden 1995); the applicant states that these methods reflect current practice. However, whales have not appeared to avoid research vessels in the past when their engines were turned off, occasionally passing directly underneath the boat (Shelden 1995). Conspecifics near target individuals also react strongly to initial vessel approaches at 10-20 m away; of 93 individuals targeted for tagging by Shelden (1995), 77 other whales reacted strongly to vessel approach. These responses seem typical of beluga whales hunted in other areas (Shelden 1995). Mothers and calves do not appear to separate during vessel

approaches (Shelden 1995).

Beluga whale response to approaches is unique among listed cetaceans in that they display a severe avoidance response to vessels approaching closely, including rapid swimming, evasion, and extended dive duration. This response is understandable from the whale's perspective, as the population has been exposed to subsistence hunting from closely approaching small vessels for decades; an individual choosing not to avoid close approaches can easily experience lethal outcomes. Exposure to such a stressor raises concerns about the degree of stress that individuals experience and the physiological, pathological, reproductive, and survival impacts, either by acute or chronic stress (Chrousos and Gold 1992; Clark et al. 2006; Cowan and Curry 2008; Waples and Gales. 2002).

As discussed, the mammalian stress response through the HPA axis is conserved in marine mammals and is the physiological mechanism by which physiological and pathological impacts of stress occur (Curry 1999; St. Aubin and Geraci. 1990; St. Aubin et al. 1996). To evaluate the significance of the stress response induced by close vessel approaches, we evaluated available data on HPA axis components in beluga whales and other taxa during baseline activities and during activities similar to or more invasive than those proposed by the NMML. For example, Schmitt et al. (2010) measured cortisol and aldosterone levels (two hormones associated with the HPA axis that change within minutes of a stressful event) in captive beluga whales during in-water blood draws as well as during more invasive out-of-water physical exams. Such procedures are a normal aspect of captive beluga whale husbandry and do not seem to impact their survival, growth, or reproduction in captivity; captive individuals are suggested to have lower stress levels due to lack of many factors individuals must deal with in the wild (Schmitt et al. 2010).

St. Aubin et al. (2001) measured levels of these same hormones in 183 free-ranging beluga whales exposed to an average of 15 min of approach followed by capture, transport to shallow water, and blood draw (activities more invasive than those proposed, but for which close approach was similar). Comparison of hormone levels indicate that animals in the free-ranging study experienced cortisol levels at blood draw roughly 150% of baseline levels in the captive study and aldosterone levels roughly an order of magnitude greater. Cortisol levels during and subsequent to the captive out-of-water exam were similar to or higher than those found in free-ranging, approached, captured, and handled individuals; aldosterone levels were 50-300% higher, though, and glucose levels were similar (glucose levels become elevated during a mammalian stress response). In a separate study, St. Aubin and Geraci (1989) chased, live captured, and sampled several dozen mostly subadult beluga whales (all but one within 30 minutes). Mean cortisol levels were almost identical to those observed in St. Aubin et al. (2001), while aldosterone levels averaged one-third less. Glucose levels were less than one-third of those found in St. Aubin et al. (2001).

For both St. Aubin et al. (2001) and St. Aubin and Geraci (1989) studies, maximum aldosterone levels were two to five fold greater than those found in the captive study by Schmitt et al. (2010). Maximum cortisol levels in Schmitt et al. (2010) were greater than those in either St. Aubin et al. (2001) or St. Aubin and Geraci (1989), but by less than two-fold. Orlov et al. (1991) also found cortisol levels of chased, captured, and captively-held beluga whales to be near the mean values of both St. Aubin et al. (2001) and St. Aubin and Gercai (1996). Norman et al. (2011)

found that glucose levels in beluga whales approached, and captured (total combined duration of 50-129 minutes, mean of 80) were not elevated, although serial sampling through to release was not performed.

Similar findings of cortisol and aldosterone responses have been made for bottlenose dolphins in the wild versus captivity (Schroeder and Keller. 1989; Thomson and Geraci. 1986). Cowan and Curry (2008) summarized cortisol response to stress as less than that found in terrestrial mammals while aldosterone response was greatly increased based upon prior cetacean work (St. Aubin and Geraci. 1990; Thomson and Geraci 1986). Unfortunately, the St. Aubin et al. (2001) and St. Aubin and Geraci (1989) studies did not follow-up with additional blood draws tracking the progress of many stress response parameters. St. Aubin and Geraci (1988) did with different hormones (triiodothyronine[T3] and thyroxine [T4]) in free-ranging beluga whales exposed to close approach, capture, handling, and release (typically less than 30 minutes combined) or captive housing over several hours to weeks, finding that levels of the hormones responded only after several hours, which was expected based upon the time for response in other marine mammal studies (St .Aubin et al. 2001; St Aubin and Geraci 1988; St. Aubin and Geraci 1992). However, no other adverse effects were noted; individuals gained weight and exhibited further hormone response to chemical induction, suggesting additional latitude in the stress response without becoming distressed. St. Aubin and Geraci (1989) did observe 20-40% decreases in some immune system parameters within 2-3 hours of capture (transport of individuals occurred during this time as well). St. Aubin et al. (1996) found that dolphins remaining encircled up to four hours retained elevated cortisol and aldosterone levels.

Based upon these findings and pending additional data on response over time in beluga whales and examination for pathological changes evident from necropsy of deceased individuals (Cowan and Curry 2008), we believe that the proposed close approaches will instigate a mild to moderate stress response in Cook Inlet beluga whales less than that observed in Schmitt et al.'s (2010) beluga whales examined out-of-water, or free-ranging individuals studied in St. Aubin et al. (1996) and St. Aubin et al. (2001). Such a stress response would be temporary and lack pathology, acute or cumulative (i.e., no distress), that would impact the target individual's survival or growth. The stress response may be greater during spring, as baseline stress hormone levels have been found to be higher in beluga whales during this time (Orlov et al. 1991).

Several studies have suggested that stress can adversely impact female reproduction through alterations in the estrus cycle (Herrenkohl and Politch 1979; Moberg 1991; Mourlon et al. 2011; Rivier 1991). Komesaroff et al. (1998) found that estrus may inhibit the stress response to some extent, although several studies suggest estrus and particularly the follicular stage may be susceptible to stress-induced disruption (see (Rivier 1991) and (Moberg 1991) for reviews). Most of these studies were conducted with single or multiple highly invasive and frequent stress methodologies or chronic stress; we do not expect stressors associated with the proposed research to be nearly as stressful. Under less invasive and acutely stressful methods (but more invasive than those proposed by the NMML), Omsjoe et al. (2009b) found no impacts to the percentage of individuals with offspring the following year following chase, capture, and restraint of reindeer (ungulates in general tend to be prone to strong, potentially lethal stress responses). Overall, we do not expect reproduction to be impaired primarily due to the lack extreme stressors utilized by studies to induce adverse reproductive impacts and the acute nature of the stressors involved.

The close approach of vessels also presents the possibility that valuable acoustic information could be missed by the target individual(s) due to masking by the vessel's engines. The acoustic properties of vessels likely to be used by the applicant are similar to the frequency range utilized by target marine mammals during vocalization such that communication could be impaired (Clark et al. 2009; Dunlop et al. 2010b). Parks et al. (2010) and Anonymous (2010) found that North Atlantic right whales temporarily modify the amplitude of their calls, making them louder with increased background noise (including noise from vessel traffic), as well as shifting call frequency over longer time frames. Killer whales in high traffic areas have been found to increase call duration or call amplitude in response to increased anthropogenic noise in the marine environment (Erbe 2002b; Foote et al. 2004b; Holt et al. 2009). As a broader issue, increased anthropogenic noise in the marine environment has the potential to reduce the range over which individuals communicate, conceivably increasing calf mortality, altering ideal group or individual spacing, and making identification and selection of mates more difficult or impossible (Croll et al. 2001b). The applicant proposes to use one vessel per survey, and we do not anticipate masking will occur for several reasons. Operations would be conducted at low speed with a minimum of throttling and directional changes. Low vessel speed means that less cavitation will occur, which is the primary source of sound energy emitted by motorized vessels (Mazzuca et al. 2001; Ross 1976). Lower speed and fewer directional changes will also result in fewer changes in sound characteristics, which are believed to add to the significance of vessel noise and its impact to cetaceans. Most interactions with target individuals should be brief (less than 30 minutes) before the vessel breaks contact following photoidentification, biopsy, and/or biopsy.

Based upon few data, Hawaiian insular false killer whales do not appear to respond to close vessel approaches; individuals frequently approach vessels to engage in bowriding (Robin Baird, pers. comm.).

We would expect most listed whales exposed to close vessel approaches under the proposed permit to exhibit either no visible reaction or short-term low-level to moderate behavioral responses. Available evidence, including approaches of individuals of other species in a variety of locations, leads us to conclude there should be no strong behavioral responses to close approaches except by Cook Inlet beluga whales, all of whom we expect to strongly evade every approach within 10 m. We expect conspecifics around the target individual to also respond strongly, but for a lesser duration as they will not be pursued. The NMML has documented responses of marine mammals to its permitted activities for several years. Based upon these data, it is reasonably possible that up to 142 humpback, 43 North Pacific right whales, four sperm whales may respond to NMML close vessel approaches in the North Pacific and Arctic Oceans every year with low- to moderate level behavioral responses described above. One or a few blue whales may also respond with low-to moderate-level behavioral responses described above for baleen whales. An unknown number (we expect up to a few) of Hawaiian insular false killer whales and southern resident killer whales may also respond to close vessel approaches, also with low- to moderate-level behavioral responses. The NMML has not attempted work on humpback whales in the North Atlantic before, so monitoring data are not available to directly assess the impacts of past NMML activities here. However, we expect the number of responses to be less than for humpback whales in the Pacific Ocean, as the level of permitted activity would be less. Table 22 on page 113-114 described the extent to which individuals may be re-exposed to proposed activities, including vessel approach. We expect that some, but not all, individuals

may respond to these re-exposures as well.

Tagging

Partially and fully implantable/dart/dash tags

Although external transmitting devices have been used by many researchers, few studies examine the possible effects of these devices (Culik et al. 1994; Hawkins 2004b; Murray and Fuller 2000; White and Garrot 1990; Wilson and McMahon 2006). For example, Murray and Fuller (2000) surveyed a sample of articles in which vertebrates had been marked, covering nine journals that publish studies on a broad range of taxonomic groups, and found that in most instances (90 percent of 238 articles surveyed), the articles did not address potential effects of marking, or at least did not report that such effects had been considered. However, the attachment of a device has the potential to generate physiological and behavioral effects, depending on factors such as device weight, shape, and attachment location (Hawkins 2004b; White and Garrot 1990). Effects of attached devices may range from subtle, short-term behavioral responses to long-term changes that affect survival and reproduction; attached devices may also cause effects not detectable in observed behaviors, such as increased energy expenditure by the tagged animal (White and Garrot 1990; Wilson and McMahon 2006). Walker and Boveng (1995) concluded the effects of devices on animal behavior are expected to be greatest when the device-to-body size ratio is large. Although the weight and size of the device may be of less concern for larger animals such as cetaceans, there is still the potential for significant effects; for example, behavioral effects that may cause reduced biological performance, particularly during critical periods such as lactation (Walker and Boveng 1995; White and Garrot 1990).

Once target individuals are approached, researchers propose to place devices in whales to track movements and dive data. This involves implanting tags into target individuals, a process that has been shown to not only result in behavioral responses, but has the potential to induce physiological and pathological changes. Implantable tags can cause behavioral responses similar to close approach as well as wounds, bruising, swelling, hydrodynamic drag, and in at least one case, lead to death. Some species are more behaviorally responsive than others, as shown in Table 23. Humpback whales tend to be one of the least responsive baleen whales to the tagging process. Although less data are available for bowheads, their closely related kin, the southern and North Atlantic right whales, tend to be amongst the most responsive species to tagging. Available data regarding the effects of tagging is almost exclusively focused on short-term effects, as few studies have attempted to follow up on tagged individuals weeks, months, or years after tagging. However, some opportunistic resightings have been documented; results are presented when available.

Table 23. Number of whales tagged and number of whales responding to the tagging process, by species (Mate et al. 2007c).

Whale species	# tagged (including failed attempts)	# responding (including failed attempts)	% responding (including failed attempts)
Blue	146	22	15.1

Fin	29	13	44.8
Humpback	122	37	30.3
North Atlantic right	16	12	75
Southern right	23	15	65.2
Sperm	60	51	85

Physiological risks to whales from tagging include swelling, inflammation, or infection of the tag site. Although concerns about the potential to strike an animal in sensitive areas, such as the eyes or blowhole, have been raised in previous studies (Whitehead et al. 1990), methods adopted by the researchers here would prevent such occurrences. To minimize localized infection risks, the parts of the tags that would be inserted into whales would be constructed of medical grade stainless steel, titanium, or other biologically inert materials, and thoroughly disinfected before attachment. Most infections in wildlife resulting from invasive tagging stem from the skin (Hawkins 2004a; Mate et al. 2007c). Invasive components are generally designed to minimize the potential for skin intrusion into the wound at time of tagging (Mate et al. 2007c). Although a wide variety of implantable tags have been used over the past several decades, review of available data support tags to generally produce a similar, small variety of wound patterns in North Atlantic right and humpback whales: white scar, white scar and divot, a divot and cyamids (whale lice), localized swelling, and regional swelling (up to 90 cm across and persisting for years), although roughly one in eight individuals showed no wound pattern (Kraus et al. 2000; Mate et al. 2007c; Quinn et al. 2000; Weller 2008). Follow-up monitoring shows local and regional swelling frequently occurs around the tag site following implantation in humpback and North Atlantic right whales (Mate et al. 2007c). Southern right whales appeared to generally lack swelling around implantable tags, but divots were frequently seen after tag rejection on individuals resighted after greater than one year post tagging (Best and Mate. 2007). Tags appear to be shed by 27-36 months post tagging, although some may protrude (begin to be ejected) after 75 days (Best and Mate. 2007). Divots are theorized to stem from fat cell rupture upon tag entry (Mate et al. 2007c). The physiological consequences of such responses remain unstudied, but a general response of glucocorticoid secretion and lymphocyte suppression is known to occur in whales entangled in fishing gear (Cole et al. 2006). Although gear entanglement has been shown to be potentially very debilitating or lethal to a whale, we expect the same response to be present, but at a lower level.

Expert reviewers in a workshop summarized by Kraus et al. (2000) were not concerned with the consequences of divots, cyamids, or scars. However, swelling was believed to be due either to hematoma, abscess, or an active inflammatory response to a foreign body or agent (such as bacteria), rupture through the subdermal sheath, foreign body granuloma, or benign tumor. Several reviewers had serious concerns for the potential of tags penetrating into the muscle layer, potentially introducing serious infections into muscle and expanding the infection due to shear forces at the muscle-blubber interface (Kraus et al. 2000; Quinn et al. 2000; Weller 2008). The extensive resighting history of North Atlantic right whales permits some analysis of tagging effects and, ultimately, survival rates of tagged versus untagged individuals is not discernibly different (Mate et al. 2007c). Resightings from other species, although not as extensive, has also

failed to support long-term effects at the individual level (Best and Mate. 2007; Mate et al. 2007c). The only close study of a wound after tagging was based upon a gray whale that stranded dead 18 days post tagging; although the animals was decomposed, investigators found no evidence of infection at the tag site or other findings that suggested the tag/tagging process resulted in the animal's death (Weller 2008).

Keeping implanted tags stable promotes healing, as new epithelial cells and scar tissue form around the foreign body to wall it off (Mate et al. 2007c). Researchers expect that the presence of recurved barbs on the cylinder housing should enable the tag to remain embedded for longer periods of time and be more stable in the body. However, over time, the tag would be rejected by the body and migrate out of the blubber due to possible infection, reaction to a foreign body, an irritation from motion due to body flexing, as well as mechanical stress from hydrodynamic drag on the external components of the tag (Watkins et al. 1981). The applicants state that tag rejection can take as little as a couple of weeks to over one year; this is supported by Watkins et al. (1981), Best and Mate (2007), and Mate et al. (2007c).

Apart from pathological effects, tagged marine mammals can also experience physiological effects, particularly from impaired hydrodynamics. Tags should be designed to minimize the drag experienced by the individual carrying the tag (Hawkins 2004a; Hooker et al. 2007). For example, Walker and Boveng (1995) found that average foraging-trip and nursing-visit durations were significantly greater for seals carrying time-depth recorders and radio transmitters than for seals carrying radio transmitters only. A spotted dolphin fitted with a bulky satellite transmitter was recaptured eight days after tagging in poor body condition, presumably due to the large drag effects it created (Scott et al. 1990). However, the tag designs under the proposed action minimizes drag, so as to increase attachment duration. Under the proposed actions, a variety of tags could be used; some have minimal drag potential (fully implantable, dart, and dash) because of their small external profile, while partially implantable tags likely experience greater overall drag because of their higher profile, but otherwise are designed for minimal drag. Hawaiian insular false killer whales have the smallest profile of all target species and would be expected to experience the greatest impact from any increase in drag. However, tag profile would be no more than 1% of the target individual's frontal cross sectional area and no more than 0.1% of its body weight. Drag would be considered minute when compared to the size of most target species, even as calves; the additional energy expenditure, even when considered over the course of a year, would be small in comparison to the drag created by such large animals in a highly-viscous medium. This is supported by data from Best and Mate (2007), who found that six out of seven female southern right whales birthed in their routine intervals (similar to the rate of detection of untagged individuals; (Best et al. 2005).

Blue whales. Blue whales tagged with implantable tags have immediately resumed lunge feeding following tagging in a large number of cases (Mate et al. 2007c).

Fin whales. Watkins (1981b) tagged several fin whales with relatively large radio transmitters and did not observe responses by targeted individuals to the actual tagging, although response to changes in vessel throttling or tags splashing on the water during misses were documented. It is noteworthy that closely related Bryde's whales have been documented to respond to both missed and successful tagging events with rapid acceleration and/or multiple breaching in two individuals; one returned to baseline behavior within 2-5 minutes, while the other individual took

2.5 hours to normalize (Watkins et al. 1979).

Humpback whales. Short-term, behavioral effects are also documented for humpback whales. General whale responses include no response at all, skin twitching, startle reactions or flinching, altered swimming speed and orientation, diving, rolling, head lifts, high back arching, fluking, and tail swishing (Goodyear 1981; Goodyear 1993b; Hooker et al. 2001c; Mate et al. 1997b; Watkins 1981c; Watkins et al. 1984b). Mate et al. (1998) found humpback whales to not respond to satellite tagging at all. Humpback whales responded to shallow implantable tags by turning away from the tagging vessel and undertaking short dives; and increasing their swimming speed (Goodyear 1993b). Watkins (1981b) found humpback whales in the North Atlantic to respond to tagging with startle reactions, increased swimming speed, or with no reaction at all; all responding individuals returned to baseline behavior within 15 minutes. A humpback whale was found to resume singing within 13 minutes of tagging in another case (Mate et al. 2007c). "Strong" reactions were found in only 3.3-5.6% of humpbacks tagged (Weinrich et al. 1991; Weinrich et al. 1992b). Humpback reactions can also occur to misses, possibly as a result of splashes in the water (Brown et al. 1994; Watkins 1981c). Baseline behavior appears to resume within minutes. Responses to tagging may be difficult to discern from responses to close approaches. In two studies of humpback whales off Hawaii and Alaska, no additional responses were found to approach and tagging versus approach alone (Mate et al. 1991; Watkins 1981c). Ultimately, humpback whale survival does not appear altered by invasive tagging; seven individuals tagged in Alaska 20-30 years ago have been reidentified in recent years also in Alaska (Mizroch et al. 2008).

Bowhead whales. Although multiple tagging studies have been conducted on bowhead whales, researchers did not report the effects of tagging on target whales (Heide-Jorgensen et al. 2006; Krutzikowsky and Mate. 2000; Laidre et al. 2007). However unpublished data from other permit applicants with bowhead tagging experience provide some insight. Only one individual has been resighted a few days after tagging; the individual's behavior appeared normal and there appeared to be nothing noteworthy about the tagging site. A few resightings have occurred in other studies up to 15 days post tagging; individuals appeared to be behaving normally and no evidence of swelling or infection was evident (A. Zerbini, unpublished data; NMML unpublished data, M. Jensen, unpublished data). Based upon bowhead satellite tracks, movement patterns have not appeared unusual. Tags remained functional for several months and the authors claim that this supports a lack of infection, as this would have resulted in tags being expelled or rubbed off.

Right whales. Goodyear (1993b) found North Atlantic right whales to not respond or twitch in response to tagging and then turn or dive away from the tagging vessel, although baseline behavior resumed within minutes. After Mate et al. (1997b) tagged mothers and calves, the pair remained together and calves ultimately survived, evidence that the authors use to support the supposition that tagging does not significantly alter the mother-calf relationship. One breeding event was also documented for tagged right whale (Weinrich et al. 1992b). Best and Mate (2007) used resighting data for previously tagged southern right whales to suggest that no major impact on reproductive output or short-term survival had occurred due to implanted tags.

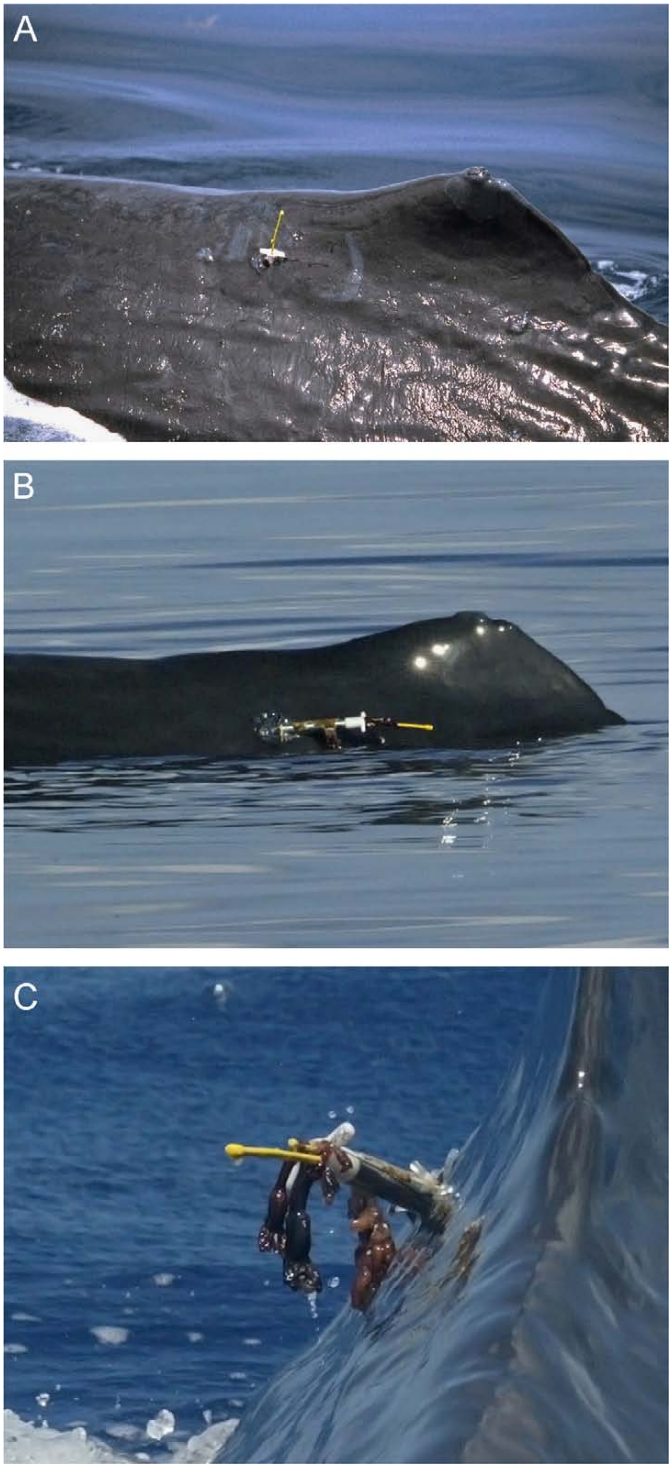

Figure 8. An implantable satellite tag being expunged from the side of a sperm whale. A. photograph of implantable tag moments after being implanted. B. and C. Same individual and tag nearly one year later. Figure from Mate et al. (2007), used with permission of Bruce Mate. Lead author notes expulsion process not likely representative of all invasive tags in all species.

Sperm whales. Responses to implantable tagging appear to vary within the species. Watkins et al. (1999) found sperm whales to not respond to tagging, including time spent at the surface,

although Watkins et al. (1993a) found a startle reaction in one individual. Tagging of seven out of ten sperm whales within a single group and within a 90 minute timeframe did not cause the group to disperse, although responses to tagging occur more in this species than any other large whale (Mate et al. 2007c). These researchers have resighted 15 of 57 tagged sperm whales, finding persistent localized swelling many months after tagging. Sperm whales tagged while resting on the surface between foraging dives appear to respond by engaging in a foraging dive earlier than they otherwise would (Tyack 2003). This dive may not last as long as it otherwise would, but conspecifics may follow the target individual in its early dive. Missed tagging attempts have resulted in a startle response (rapid acceleration and defecation), although tagging hits did not appear to elicit responses (Watkins and Tyack 1991).

Hawaiian insular false killer whales. Data from false killer whales is generally lacking, but unpublished data from Dr. Robin Baird are available to assess impacts of dart tagging. Dr. Baird has found that between 2006 and 2009, false killer whales in Hawaiian waters do not respond to dart tagging attempts in 9% of cases and respond by accelerated dives, tail flicks, and or increased swimming speed in 91% of 23 cases. These responses appear to be short-term, although follow-up monitoring is limited to observations of scarring and some tissue inflammation.

Studies of other toothed whales are also available to assess responses. Tagging has been conducted on a variety of marine mammal species, including pilot whales (Mate 1989), blue whales (2003; Calambokidis et al. 2001b; 2007; Lagerquist et al. 2000; Mate et al. 2007b), beluga whales (Martin and Smith 1992), northern bottlenose whales (Hooker et al. 2001a), Hector's dolphins (Stone et al. 1994), bottlenose dolphins (Schneider et al. 1998a), Dall's porpoises (Baird and Hanson 1996), harbor porpoises (Eskesen et al. 2009), and narwhals (Martin et al. 1994). Although several tagging studies have been conducted, few have systematically investigated or recorded the effects on cetaceans from tagging, and available investigations into instrument effects on marine species are often limited to visual assessments of behavior (Walker and Boveng 1995). In addition, reactions to tagging are difficult to differentiate from reactions to close vessel approaches, because in all cases it is necessary to closely approach the individual to ensure proper tag placement.

Southern resident killer whales. Perhaps the best data regarding response comes from killer whales in the North Pacific Ocean. Data on responses of killer whales to tagging are available from studies conducted on 32 individuals from four different lineages (Antarctic Type A and Type B, North Pacific Residents and Transients) between January 2006 and September 2007 (unpublished data from Andrews et al. cited in (NMFS 2008c)). Most of the tagged whales were adult or sub-adult males, and more than half (18) were reported to exhibit no immediate reaction to tagging. Nine whales exhibited a slight or very slight startle or shake in response to tagging. A moderate startle reaction was observed in two whales (both transients), and three whales responded to tagging by either a startle response (no magnitude noted) or a startle response combined with a roll or dive. Of these 32 whales, 13 individuals were resident ecotype killer whales. The reactions noted for the resident whales were consistent with the range of reactions noted for the larger data set encompassing all 32 killer whales (i.e., 7 no reaction, 5 slight or very slight startle, 1 startle and dive). The duration of tag attachments averaged 29 days up to a maximum of 65 days, similar to what is anticipated during the proposed study. Since this tagging work, follow-up studies in Alaska have re-photographed previously tagged individuals

over periods ranging from days to two years. Researchers report the available data from tagged whales indicate no long-term behavioral reactions and negligible scarring that is difficult to detect at the tag site (unpublished data from Andrews et al.; K. Balcomb, personal communication cited in (NMFS 2008c)). For example, photographs of whale AK1 which was tagged twice – once on August 9, 2006 and again on June 12, 2007 – indicate that tags sites completely healed, with small localized swelling that eventually subsided leaving no visible irregularities on the dorsal fin (unpublished data from Andrews et al cited in (NMFS 2008c)).

Andrews et al. (2005) also used satellite dart tags on five resident killer whales in southeastern Alaska. Tags were deployed using a similar crossbow and with similar tags as the proposed study, but with one dart for attachment and a penetration depth of 3 cm. In September 2004, three killer whales were tagged in various locations on the dorsal fin and flank; the authors observed no reactions to tagging. Three weeks later one of the tagged whales was resighted and a small (1-cm or 0.4-in) but healed spot was observed at the tag site. The authors concluded this suggested minimal tissue reaction and quick healing time. In October 2004, two killer whales were tagged and no reaction was noted from either whale. The authors used video playback to confirm the absence of an observable behavioral response to tagging. Both of these whales were followed for approximately 2.5 hours after tagging during which they exhibited apparently normal behavior. The authors also noted that no bleeding was observed when tags struck the dorsal fin.

Erickson (1978a) reported on an earlier use of attached radio packages on the dorsal fin of two killer whales in the Pacific Northwest. Although these whales were equipped with a much larger radio package (1.4 kg) than the proposed tags, information from this study informs our assessment of anticipated effects. The radio packages were affixed to the base of each dorsal fin using four surgical pins that pierced the fin. The author noted that attachment of the tag elicited no noticeable reaction from the whales, no bleeding from the pin sites, and no flinching or thrashing at any time during the attachment process. Whales were observed afterward to determine if disturbed by the package; however no behavioral aberrations were seen and the whale did not attempt to rub the transmitter or try to remove the package (Erickson 1978a). The two whales were later recaptured and examined; there was no evidence of tissue edema, skin irritation, or discharge suggestive of infection. The author also noted that blood samples showed no significant change in the blood characteristics of either whale, particularly the white blood cell count, coincident with the attachment of the radio package.

Other odontocetes have also been assessed for response to tagging. Cuvier's and dense-beaked whales exhibited fast dives and/or tail splashes upon tagging, but had returned to baseline behavior upon resighting (Baird et al. 2004). Captive common dolphins have been observed to pull on implanted tags of conspecifics; similar behaviors are suspected of bottlenose dolphins (Scott et al. 1990). A shark bite in a wild dolphin appeared to have been directed at an implanted tag (Scott et al. 1990).

Suction-cup tagging

***Baleen whales*.** Although suction cup tagging is not as invasive as implantable tagging, whales have also demonstrated behavioral reactions to tag attachment. Goodyear (1989c) observed a quickened dive, high back arch, tail swish (31%) or no reaction (69%) to suction cup attachment, although one breach was observed in roughly 100 taggings. Baird et al. (2000) also found

responses less frequently than responses in humpbacks (17% of 31 attachments), although competitive groups were easier to approach than singletons. Regardless, pre-tagging behavior was observed again in all cases within minutes. No damage to skin was found (Goodyear 1989a). Baumgartner and Mate (2003) reported that strong reactions of North Atlantic right whales to suction-cup tagging were uncommon, and that 71% of the 42 whales closely approached for suction-cup tagging showed no observable reaction. Of the remaining whales, reactions included lifting of the head or flukes, rolling, back-arching, or performing head lunges. No differences in dive patterns were found after two dives post-tagging. Suction cup tagging of bowhead whales has met with poor attachment success due to the animal's rough skin and evasive behavior (Baumgartner and Hammar. 2010).

Whether any long-term effects resulting from tagging remain largely unknown and available information is limited. Goodyear (1989b) noted that humpbacks monitored several days after being suction-cup tagged did not appear to exhibit altered behavior.

Although reported data are relatively paucious on baleen whale responses to suction cup tagging, discussions with experts having years of experience in the field provide additional insight into likely response. Overall, suction cup tagging produces similar responses as biopsy or more invasive tagging, with low-level, ephemeral responses or no response observed in most cases (David Schorr, Cascadia Research, pers. comm.).

Southern resident killer whales. Several studies using suction-cup attached tags on killer whales are available. These studies do not involve the use of invasive tags; however, whales are documented to respond to the strike of a tag and the results of these studies are also informative for this consultation. Baird (1994) attached suction cup TDR tags to three killer whales using a pole, noting that two reacted with a low-level response (flinch and roll) and one by swimming away. Using crossbow deployment, the author also tagged seven killer whales and noted no reaction in 43% of the whales and a low-level reaction in 57%. The author reported that whales were not more difficult to approach after tagging than before, and suggested their behavior was not greatly modified due to tagging. Baird (1998) reported on additional tagging studies using a crossbow-deployed suction-cup tag on killer whales. Of over 160 attempts (41 successful), the author noted that responses were either no reaction or low-intensity and short duration responses such as flinching. Baird et al. (2003b) tagged eight southern resident killer whales with suction-cup tags and characterized killer whale behavior during the study as including social and travel behaviors; however, no description of behavioral or other responses to tagging were noted. Baird et al. (2005b) reported on tagging studies of 34 southern resident killer whales using a suction-cup TDR tag. Males between 3–42 years of age and females between 3–60 years old were tagged at distances of approximately three to seven meters from the whale, using a crossbow. Immediate reactions included no reaction (24% in U.S. waters from 1997-2002) and low-to-moderate behavioral reactions consisting of a fast dive and a flinch or tail flick (76% in U.S. waters 1997-2002). No strong behavioral reactions were observed, and no changes in general behavioral state (e.g., travel, foraging) were seen immediately following tagging (Baird et al. 2005b). In addition, acoustic monitoring of one event documented no change in sound production associated with the tagging.

No research has been done to specifically assess the long-term impacts of tagging on killer whales. However, as described above, data from resightings of previously tagged killer whales in

Alaska days to years after tagging suggest no long-term behavioral reactions or physical damage (unpublished data from Andrews et al. K. Balcomb, personal communication cited in (NMFS 2008c)).

Cook Inlet beluga whales. Suction cup tagging of Cook Inlet beluga whales does not appear to elicit a response different from those described in the process of approaching target individuals (Lerczak et al. 2000).

Hawaiian insular false killer whales. As with implantable tagging, few data are available on false killer whales, but suction cup tagging has been attempted on other small odontocetes. Roberts et al. (1999) found a false killer whale to respond to a suction cup tagging attempt with a fast dive, but did not subsequently avoid the research vessel. A conspecific subsequently pulled the tag off after one hour. This study also documented spotted and spinner dolphins to respond to tagging with tail flicks or fast dives, but most animals returned to the research vessel to bowride. Hanson and Baird (1998) found bowriding Dall's porpoises to react in 11 of 13 successful tagging events, but in none of the two misses. Responses included tail slaps, flinches, and/or swimming rapidly away. However, in seven of 11 responses, individuals returned to bowriding and telemetry data suggest individuals returned to baseline behavior within eight minutes. Northern bottlenose whales generally do not respond to missed tagging attempts and usually respond to hits by low to moderate-level reactions, but returned to baseline behavior within minutes (Hooker et al. 2001b). Bottlenose dolphins appear to respond very strongly to suction cup tagging, engaging in immediate and continuous leaping and increases in swimming speed in nearly all cases (Schneider et al. 1998b). Stone et al. (1994) found a single successful tagging event on a Hector's dolphin caused the individual to cease bowriding and depart the area, but return to bowride within five minutes.

Data from NMML monitoring reports are not entirely clear as to the number and type of responses target individuals show to tagging activities, nor the type of tagging conducted. Due to this lack of detail, we could not identify the number of individuals tagged using invasive means versus suction-cup tagging over the past several years of NMML's activities. However, review of the literature and discussion with experts supports responses and response rates by target species to be generally similar to these forms of tagging. Response data provided by Mate et al. (2007a) for blue, fin, humpback, right (used as a surrogate for bowhead whale response due to lack of species-specific response data), and sperm whales appears to be the best source to appraise the rate of response by these species to invasive tagging and is used here to estimate the number of responses; additional information summarized above helps us determine the type of response likely to occur under the proposed permit. Baird (2011, pers. comm.) provides a basis for estimating response rate of Hawaiian insular false killer whales (91%). Based upon these response rates and the expected level of tagging, we do not expect sei whales to respond, but two fin, eight humpback (Pacific), 25 bowhead, six North Pacific right, and 11 sperm whales are expected to respond to invasive and suction cup tagging activities with low- to moderate-level behavioral responses described above. One or a few blue whales may also respond in a similar manner over the life of the proposed permit. As it is possible that an individual could be exposed to tagging more than once per year, the same individual could respond multiple times (potentially as frequently as the maximum annual range identified in Table 22 on pages 113-114). In addition, 167 humpback whales (Atlantic), most if not all Hawaiian insular false killer whales, and most if not all of the southern resident killer whales are expected to respond to tagging

(again, multiple individual responses are possible). We do not expect Cook Inlet beluga whales to respond to suction cup tagging (at least not in a way that is discernable from stressors associated with close vessel approach)(Lerczak et al. 2000).

Most responses would consist of low-level, transitory behavioral responses, such as startle, flinching, defecation, fluke beat(s), premature or accelerated dive, movement away from the research vessel, increased swimming speed, rolling, head lifts, and/or back arching. Some individuals may exhibit more prolonged or extreme responses, rising to a moderate level. We do not anticipate any strong behavioral responses to tagging. We expect all individuals receiving implantable tags to experience a physiological response to the foreign body, including swelling or inflammation. We do not anticipate any individual will incur an infection from tag application, although data are spartan and additional study is needed to better inform this possibility.

Our use of behavior as an indicator of a whale's response to tagging may or may not accurately reflect the whale's experience, and we cannot definitively know whether such behavioral responses have long-term consequences. Responses to human disturbances, such as tagging, may manifest as stress responses, interruptions of essential behavioral or physiological events, alteration of an animal's time budget, or some combination of these responses. Weinrich et al. (1992) associated "moderate" responses with alarm reactions and "strong" behavioral reactions with stress responses. Wild harbor porpoises restrained and tagged did not show consistent elevations in cortisol nor did heart rate change in ways consistent with a stress reaction (Eskesen et al. 2009); these actions are much more invasive that those proposed. Moderate responses might also be associated with a stress response, given that certain behavioral responses may have metabolic consequences. As a result, we assume the proposed tagging could be stressful for a small portion of the whales; however, the significance of this stress response and its consequences, if any, on the fitness of individual whales are not definitively known. However, the limited information available from Erickson (1978a) indicates that for a more invasive radio package attachment on the dorsal fin, the blood parameters of killer whales showed no significant change. Recognizing the evidence indicating that behavioral responses would be short-lived, we provisionally assume that the tagging activities could produce short-lived stress responses in some individuals.

Biopsy

Biopsy sampling has the potential to disrupt behavior and breach an individual's integument. Physiological, pathological, and behavioral responses are possible. We reviewed the literature assessing the impacts of biopsy sampling to various cetacean species. We know of only one published report of a cetacean death following biopsy sampling, when the dart penetrated the muscle mass of a female common dolphin (*Delphinus delphis*), which may have resulted in vertebral trauma and severe shock (Bearzi 2000). The individual had relatively thin blubber, permitting deeper penetration than was desired and sticking of the dart. Apart from the one mortality, there is not even evidence of infection at the point of penetration or elsewhere among the many whales sighted in the days following biopsy sampling (Weller 2008). The risk of infection is thought to be minimized by sterilizing dart tips before sampling occurs. In general, healing is rapid (roughly one week, scarring thereafter)(Noren and Mocklin 2011).

Balaenopterids. Although suction cup tagging has become a common field method for studying

baleen whales, few data exist regarding response of balaenopterid whales to suction cup tagging. Gauthier and Sears (1999) summarized data for several species, including blue, fin, and humpback (Table 24). Blue whale responses responded by submerging, accelerating, and/or diving (Gauthier and Sears 1999). Fin whales either do not respond at all, or exhibit low- to moderate-level behavioral responses (Marsili and Focardi 1996). Inadvertent repeated biopsy within a week did not appear to cause a difference in reaction in three blue whales and five fin whales (Gauthier and Sears 1999). Group size does not appear to impact the likelihood or severity of response (Gauthier and Sears 1999). Female fin whales appear to respond to biopsy more often than males (66% versus 44%) and more strongly. Individuals generally return to baseline behavior within a few minutes (Gauthier and Sears 1999). A biopsy miss that hit the water near a target fin whale apparently caused the fin whale to dive (Gauthier and Sears 1999).

Table 24. Response frequency and intensity of baleen whales to biopsy attempts (Gauthier and Sears 1999)

Whale species	Response	No response	% responding	Low-intensity response	Mid-intensity response	Strong-intensity response
Blue	32	71	31.1	25	7	0
Fin	57	56	50.4	34	23	0
Humpback	135	71	65.5	38	87	10

Humpback whale. Many researchers claim that biopsy darts or sampling does not result in significant short-term or long-term behavioral disturbance to humpback whales. However, humpback whales do appear to be more reactive to biopsies than other baleen whale species (Table 24). An IWC working group reviewed biopsy sampling and concluded long-term effects are unlikely, although short-term responses frequently occur (IWC 1991). Clapham and Mattila (1993) found 44% of humpback whales sampled showed no immediate response, while 22.5% reacted in subtle or minor ways. Cerchio (2003) found similar results in 350 biopsy events. Cantor et al. (2010) found that 46% of 542 biopsy attempts on adult or subadult humpback whales from 10-25 m away resulted in a behavioral response (most commonly fluke movement). Neither the use of a tether, the duration of vessel contact with the target individual, nor region of the body hit influenced the likelihood of response, although responses were more frequent and intense from smaller vessels (likely due to their additional noise) than from larger vessels. Weinrich et al. (1991) reached the same conclusions for humpback whales, although short-term disruption of foraging could occur as well as agonistic behavior and altered dive parameters. Gauthier and Sears (1999) found humpback whales to accelerate, change direction, dive, lobtail, exhale forcefully, submerge, and display tail and flipper movements (the most common response); “moderate” responses were the most common category of response. Weinrich et al. (1992b) also found that of 71 humpback whales biopsied, 7% had no response, 27% exhibited a “low” response, 61% had a “moderate” response, and 6% had a “strong” response. Brown et al. (1994) found 41% of 203 humpbacks biopsied to respond in some way, including fluke movements, tail slaps, and disrupted dives. Humpbacks rarely display tail flicks, but frequently do so in response to biopsy (Weinrich et al. 1992b). Repeated sampling was not found to

influence the likelihood of subsequent biopsy responses (Brown et al. 1994).

The behavioral state of individuals pre-biopsy may also influence the probability of response, with foraging, traveling, or socializing individuals less likely to respond than resting individuals (Cantor et al. 2010; Weinrich et al. 1991), although this is confounded by data in other areas, possibly due to differences in vessels or methods used between studies (Brown et al. 1994). Clapham and Mattila (1993) found that evasion was the most common behavioral change and that response was less likely on breeding grounds. Unlike close approach, demographic factors do not appear to influence biopsy response in humpback whales; individual age, gender, group size, geographic location, and repeated sampling have not been found to influence the likelihood of biopsy responses (Cantor et al. 2010; Gauthier and Sears 1999; Weinrich et al. 1991). Brown et al. (1994) did find females to respond more frequently than males, although not significantly so. Of individuals that do respond, return to baseline behavior occurs within a few minutes (Gauthier and Sears 1999). Mothers and males in competitive groups reacted less frequently than other individuals (Cerchio 2003; Clapham and Mattila 1993). However, calves tend to be more evasive than any other group. Females with calves responded more frequently than did non-lactating females (60% versus 43%)(Cantor et al. 2010).

Biopsy misses can also cause behavioral responses (Gauthier and Sears 1999). Strong behavioral responses were found by Weinrich et al. (1992b) and (1991) when a line attached to the biopsy dart snagged on an individual's flukes. Brown et al. (1994) reported that 16% of missed Australian humpbacks responded, suggesting that these animals reacted to the sound of the dart hitting the water. Similarly, Clapham and Mattila (1993) reported that a total of 375 (87.7%) of misses on breeding grounds involved no reaction. Gauthier and Sears (1999) found four out of five misses of individuals in a feeding area did not involve a response, although four out of five other individuals did respond until freed from biopsy darts that stuck in their blubber. Significantly stronger reactions were displayed when biopsy darts actually hit humpback whales than when they missed (Weinrich and Kuhlberg. 1991).

Bowhead whale. Prior experience by researchers has found that bowhead whales biopsied using a crossbow, from a boat, tend to dive 52% of the time, have no observed reaction on 19% of biopsy samples, or increase swimming speed/change direction on 19% of occasions (Rossi 2009). However, the reaction to the boat is usually more pronounced than the reaction to the biopsy. This is the limit of our knowledge of bowhead responses to biopsy sampling.

Right whales. The relatively high level of behavioral responsiveness observed in bowheads also appears to be present in right whale species. North Atlantic right whales showed immediate, minor behavioral response to biopsy darting 19% of the time in 241 attempts and no reaction in 81% of hits and misses (Brown et al. 1991b). Reactions include twitches, increased swimming speed and dives, back arches and dives, tail flicks, lobtails, and turning away from the tagging vessel (Brown et al. 1991b). More than 50% of individuals had a hard tail flick; an unusual behavior for this species. Dives also became longer relative to surface times. However, return to baseline behavior generally occurred rapidly (Brown et al. 1991b). It should be noted, though, that one individual lobtailed for 40 minutes after a missed biopsy attempt where monofilament line attached to the arrow trailed after the animal (Brown et al. 1991b). Reeb and Best (2006) also documented generally no or low- to moderate-level responses of right whales to pole biopsy techniques. Demographic differences in responses have been identified in southern right whales,

with greater response in singletons versus groups and cow/calf pairs responding more strongly than other groups (Best et al. 2005). Overall, changes in reproductive output by female right whales was not found, although the power to detect differences was low (Best et al. 2005).

Sperm whales. We identified only one study that has reported on the response of sperm whales to biopsy attempts. Whitehead et al. (1990) reported responses from sperm whales off Nova Scotia as well as the Azores, finding that every biopsy hit and roughly half of the misses caused a startle response. Startling was associated with flexing the body, raising the back, and/or increasing swimming speed. Other responses occasionally observed included short dives of up to five minutes and defecation. In all cases, individuals were observed to return to baseline behavior within minutes. Discussions with experienced field biologists suggest these trends are generally accurate, although no response may also occur to biopsy hits (Greg Schorr, Cascadia Research, pers. comm.).

Southern resident killer whales. Killer whales normally flinch, shake, and/or accelerate in response to biopsy hits (81%) and misses (53%), but do not show aversion to reapproaches in most cases shortly after biopsy (Barrett-Lennard et al. 1996).

Cook Inlet beluga whales. We are not aware of data documenting the response of beluga whales to biopsy attempts. We expect that, as with suction cup tagging, responses could not be discriminated from the response previously described to close vessel approach.

Hawaiian insular false killer whales. Few data are available from false killer whales. However, from other odontocete species are available. A total of 28 responses (23 low-level, five moderate) out of 49 biopsy events were measured for northern bottlenose whales, with logging individuals responding more strongly than milling or traveling individuals (Hooker et al. 2001b). Responses occurred to both hits and misses, although hits elicited much more frequent responses. Most responses were startles. Weller et al. (1997) found that all eight bottlenose dolphins biopsy sampled exhibited startle responses in his study, including tail flicks, increased swimming speed, and leaving the area. Misses did not produce a response. Follow-up veterinary and pathological examination found wounds to be uninfected and healing well, with skin covering the wound 15-42 days post-biopsy. Biopsies of roughly 100 bottlenose dolphins and four humpbacked dolphins generally resulted in startle responses as well (Weller et al. 1997). Long-term effects have not been found to result from biopsy attempts (Weller et al. 1997). Killer whales have been observed to shake or accelerate upon biopsy, but no other effects were observed (Barrett-Lennard et al. 1996). Resting and socializing groups of spinner, pantropical spotted, melon-headed, and Indo-Pacific bottlenose dolphins apparently respond more strongly than do milling or traveling groups (Kiszka et al. 2010b). Responses were similar for both hits and misses. Overall, 94% of responses were twitch and/or dive, while 2% of responses included tail slap, leaping, multiple breaches, and/or escape. Group behavior changed in 54% of biopsy attempts, with group dive being the most common response. However, group escape or increased swimming speed occurred in 4% of biopsy attempts. Group size did not bear of the likelihood or strength of response. Gorgone et al. (2008) found that 22% of conspecifics reacted in a manner similar or identical to target individuals.

As with tagging activities, NMML annual reports are unclear as to the number and types of responses target individuals exhibited upon biopsy. Therefore, we relied upon available literature and expert opinion to determine the number and types of responses under the proposed

activities. Gauthier and Sears (1999) provide the only quantitative data available for balaenopterid response, as does Whitehead et al. (1990) for sperm whales. Humpback whale responses have been documented extensively. Of the available studies, Cantor et al. (2010) and Brown et al. (1994) provide the largest sample sizes and report similar response rates; we use these studies to determine humpback response rate and the entirety of the literature to inform the expected type of response. Data from Rossi (2009) are used to calculate bowhead response rate and Brown et al. (1991a) was used for right whales. Overall, we do not expect sei whales to respond to biopsy, but 18 fin, 1,019 and 35 humpback (Pacific/Atlantic), 21 bowhead, two North pacific right, and 35 sperm whales are likely to respond behaviorally to biopsy activities as described above (mild- to moderate-behavioral responses). We also expect that one or a few blue whales may respond with low- to moderate-level behavioral responses over the life of the proposed permit. We do not expect Cook Inlet beluga whales to respond in a way that is discernable from the previously described response to close vessel approach. Based upon data from a variety of odontocete species, we expect most if not all Hawaiian insular false killer whales to respond to biopsy attempts. We could not assess the impacts of biopsy and tagging independently from one another for populations or species which we provisionally accepted proposed levels of tagging and biopsy (humpback whales in the Atlantic, southern resident killer whales, and Hawaiian insular false killer whales). However, as response rates and response types for these activities are generally similar and if biopsy and tagging were to occur to the same individual, the response to both activities by an individual would likely be the same as to one of the activities alone. As previously mentioned, individuals re-exposed to proposed activities could also undergo additional responses. Table 22 on pages 113-114 provides maximum expected re-exposure; we expect the number of individual responses to be less than this.

We expect responses to consist of brief, low-level to moderate behavioral responses, consistent with findings of Noren and Mocklin (2011). These are likely to include increased swimming speed, diving, change in direction, lobtail, forceful exhalation, submergence, tail and flipper movements, agonistic behavior, twitches, back arches, and defecation. As a result, individuals may temporarily leave the area or cease feeding, breeding, resting, or other activities. However, we expect that individuals would return to baseline behavior within a few minutes.

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 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.

This section attempts to identify the likely changes present in the future and their impact on ESA listed species and their critical habitats in the action area. This section is not meant to be a comprehensive socio-economic evaluation, but a brief outlook on future changes on the environment. Projections are based upon recognized organizations producing best-available information and reasonable rough-trend estimates of change stemming from these data. However, all changes are based upon projections that are subject to error and alteration by complex economic and social interactions. It is reinforced that projections are broad-scale and do not incorporate small- to medium-sized changes on the local level. Information sources include the U.S. Census Bureau, Department of Labor, and Lexus-Nexus information system. With the later (source for state legislation), only pending bills under consideration were included; those that

died in process or were vetoed are not included.

Regulatory changes alter the environment under which future actions can occur, including zoning regulation, fishery capacity, environmental standards, and development of commerce and industry. New York State legislation would prevent the release of balloons due to marine life ingestion. A bill in the Virginia house and senate would limit pound net fishing. Florida's legislature is considering a possible amendment to the state constitution prohibiting marine oil exploration, drilling, and extraction.

The NMFS expects whale watching operations, vessel traffic, climate change, and research activities to continue within the range of the species for the foreseeable future. The best scientific and commercial data available provide little specific information on any long-term effects of these potential sources of disturbance on whale populations. Harvests of bowhead whales in the Arctic is expected to continue in both Russian and Alaskan waters. Information on the effects of repeated harassment by research activities, vessel traffic, and whale watchers is also lacking. Lusseau (2004) provides evidence that dolphins may be changing their activity budget and behavior in response to dolphin watching tours. Salden (1988) and Herman (1979) demonstrated a shift in humpback distribution possibly due to recreational watercraft. There is also some concern that the increasing population trends for these species may be related to increased sampling effort (Branch 2006) or habitat shifts from one region to another globally, so that the global population is not increasing but the local populations at some feeding or breeding sites is. Therefore, without additional information on their population structure, which is provided by this research, continuation of these activities does not appear to pose any threat to, or prevent the survival and recovery of listed marine mammals.

States along the Pacific coast, or which contribute water to major river systems here, are projected to have the most rapid growth of any area in the U.S. within the next few decades. This is particularly true for coastal states and those of the desert southwest. California, Oregon, Washington State, Arizona, Idaho, Utah, Nevada, and Alaska are forecasted to have double digit increases in population growth rates for each decade from 2000 to 2030 (USCB 2005b). Overall, this region had a projected population of 65.6 million people in 2005 and will likely grow to 70.0 million in 2010 and 74.4 million in 2015, making it by far the most populous region (but also containing the greatest land area). The U.S. Census Bureau projects the population of Washington State (the vast majority of which lives along Puget Sound, the San Juan Strait, or coastal Pacific waters where southern resident killer whales occur) is growing at an accelerated rate of 1.1% annually by 2010, 1.4% between 2010 and 2020, and 1.6% between 2020 and 2030 (USCB 2005a; USCB 2005b). Oregon should experience similar, although slower growth. Specifically, NOAA's State of the Coast which summarizes United States census data for coastal regions, indicates that all counties within the Washington and Oregon coastal watershed will show significant increases in population, with counties along the southern resident killer whale critical habitat having some of the largest growth (http://stateofthecoast.noaa.gov/). Washington's Office of Financial Management estimates an additional 700,000 people will be living in the Puget Sound Region over the next 10 years. With this growth comes the related concern of increased toxic runoff and increased hard surface area that facilitate the runoff, reduced oxygen levels in the sound due to waste discharge, loss of habitat, and increased shoreline development. Growth in the region will increase contaminants from wastewater treatment plants and sediments from sprawling urban and suburban development that enter riverine, estuarine, and marine habitats. Environmental contamination is a persistent and long-

term health risk for some listed species, such as southern resident killer whales (Krahn et al. 2007; NMFS 2008e; Ross et al. 2000c). Overall, exposure of southern resident killer whales to most contaminants in the action area is not expected to appreciably decrease in the foreseeable future (Grant and Ross 2002; Krahn et al. 2002a; NMFS 2008e).

Whale-watching activities likely will become more extensive, potentially altering marine mammal habitat use and risk behavioral changes similar to what has been observed in the Pacific northwest (NMFS 2008e). Whale watching can become more extensive locally (i.e., Virginia Beach, a popular tourist destination roughly 80 miles from Richmond, VA, is a location past which humpback and right whales migrate) and potentially present issues with overuse.

Nationwide, construction is forecasted to be one of the most extensively growing industries in the U.S. From 2006 to 2016, the construction industry is expected to grow by 1.4%/year and employ an additional 600,000 people during that time (Figueroa and Woods 2007). However, this represents a 30% slow-down from the 1996 to 2006 time period. Construction will be most likely to occur in school, industrial, and medical areas, as well as infrastructure (bridge and road) repair and replacement. An increase in construction will entail additional development in urban and non-urbanized areas that can introduce large amounts of sediment into waterways via run-off. Sediment run-off can also introduce nutrients into marine environments that can cause algal blooms, which have been documented in nearshore habitats of the northeast U.S., and introduce neurotoxins to large areas and cause wide-scale mortality (NMFS 2007a; NMFS and USFWS 2004; Vitousek et al. 1997).

Fishing is a major industry affecting the action area. According to projections from the U.S. Department of Labor, fishing is expected to undergo decline in output (USBLS 2010a; USBLS 2010b). This means that the adverse affects of ship-strikes, entanglement, and/or competition for prey that are associated with this industry to baleen whales, southern resident killer whales, and Steller sea lions are likely to decrease.

Any future scientific studies targeting these species and contributing to their conservation or recovery will require consultation under the ESA and such studies, therefore, are not included in the Cumulative Effects section of this Opinion. Whale watching has the potential to significantly expand, generating an additional 413 million dollars globally and generating an additional 5,700 jobs (Cisneros-Montemayor et al. 2010). Similarly, alternative energy projects such as ocean current, wave, and tidal energy projects are expected to increase within the action area and will require federal permits (e.g., Federal Energy Regulatory Commission (FERC)), so these actions are not included in the Cumulative Effects section of this Opinion. After reviewing available information, NMFS is not aware of effects from any additional future non-federal activities in the action area that would not require federal authorization or funding and are reasonably certain to occur during the foreseeable future.

The state of Washington has implemented a strategy to restore Puget Sound to a healthier condition in 2020. A Puget Sound Partnership was created by the governor and in 2008 the Puget Sound Action Agenda was released by the partnership. During the legislature's 2009 session, $78.5 million dollars was earmarked for various projects that would support the action agenda, indicating a concerted effort by the state to improve the Puget Sound environment.

Integration and synthesis of effects

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

The NMFS Permits Division proposes to issue a permit amendment to Dr. John Bengtson of the NMML for directed take of all listed and proposed for listing cetaceans in the North Pacific (blue, fin, sei, humpback, bowhead, North Pacific right, sperm, Cook Inlet beluga, southern resident killer whales, and Hawaiian insular false killer whales) as well as humpback whales in the North Atlantic, all of whom are endangered throughout their ranges (except Hawaiian insular false killer whales, which are proposed as endangered). The Permits Division also proposes to authorize unintentional take for both DPSs of Steller sea lions (eastern DPS listed as threatened, western DPS listed as endangered), the Beringia DPS of bearded seals (proposed as threatened), as well as the Arctic DPS of ringed seals (proposed as threatened).

The *Status of listed resources* section identified commercial whaling as the primary reason population sizes are a fraction of their former abundance for large whales. Native harvests, strandings, and killer whale predation were identified as reasons for decline of Cook Inlet beluga whales, while collections for aquaria have been a major contributor to a reduced southern resident killer whale population size. Steller sea lions have and continue to experience a variety of threats, including fishery interaction, hunting, predation, and prey depletion. Bearded and ringed seals are not known to have undergone reduced population sizes, but are threatened by reductions in ice habitat. Insular Hawaiian false killer whales have likely experienced reductions due to fishery interactions and may be negatively impacted from environmental contaminants and prey depletion. Other worldwide threats to the survival and recovery of listed marine mammals include ship strike, entanglement in fishing gear, and toxic chemical burden and biotoxins. Listed marine mammal populations in the North Pacific and North Atlantic are expected face area-specific threats identified in the *Environmental baseline*, including habitat degradation, whale watching, research activities, seismic surveys, oil and gas exploration, naval activities, climate change, human noise sources, ship strike, subsistence and commercial harvesting, and entanglement. Despite these pressures, most large whale populations (as well as the eastern Steller sea lion DPS) appear to be recovering, although Cook Inlet beluga whale, southern resident killer whale, and the western Steller sea lion DPS population have recently fluctuated and are mostly in decline. Insular Hawaiian false killer whales appear to be either stable or in decline. Beringia bearded seal and Arctic ringed seal population sizes are poorly known, but appear to be stable. Reasonably likely future actions described in the *Cumulative effects* section include the continuation of activities previously identified in the *Environmental baseline* as well as state regulatory trends, population growth, and increase in some industrial sectors that can

degrade habitat quality.

The *Exposure analysis* describes the actions proposed to be undertaken to North Pacific populations of listed or proposed for listed and proposed for listing marine mammals: close approaches by survey aircraft of any age/sex, close approaches by research vessel of any age/sex, invasive tagging, and/or suction cup tagging (see Tables 21 and 22 on pages 110-112 and 113-114 for expected exposure and Table 20 on pages 107-109 for age class limitations on some tagging and biopsy activities).

The *Response analysis* considered that stressors to which targeted individuals would be exposed will likely cause behavioral, physiological, and displacement responses. Aerial surveys are expected to cause temporary, low-level behavioral responses in a few individuals of some listed species which are targeted under the proposed permit. We expect the vast majority of individuals to not respond at all and the remainder to exhibit low-level behavioral responses and possibly a mild stress response. Those that do respond are expected to return to baseline behavior within minutes and no targeted individual will experience a reduction in growth, reproduction, or survival potential.

Vessel approaches frequently result in behavioral changes in listed whales, with most approaches resulting in no response or apparently “minor” to “moderate” responses (increasing swim speed and direction, startle reaction, movement away, changes in respiration and diving, agonistic behavior, evasion (Baker et al. 1983b; Baker and Herman. 1989; Bauer and Herman 1986; Bauer 1986; Clapham and Mattila 1993; Hall 1982; Hemphill et al. 2006; Koski and Johnson 1987; Malme et al. 1983b; Malme et al. 1984b; Richardson et al. 1985b; Scheidat et al. 2004)). There is also compelling evidence that cumulative vessel approaches (Bauer and Herman 1986; Herman 1979) or additive effects of vessel approach and other anthropogenic stressors (Fraker et al. 1982) can have more significant effects, including the displacement of humpback whales from Alaskan foraging areas, displacement of gray whales from habitat (Reeves 1977) and interference with bowhead feeding and social behavior (Bauer and Herman 1986). The presence of additional anthropogenic stressors, such as seismic survey and drilling activities and commercial vessel traffic are likely to induce additional disturbance on potential target individuals (Fraker et al. 1982; NMFS 2008a). Although it is possible that individuals are being displaced from more preferable habitat, we have no evidence to suggest this. The number and severity of responses to vessel approaches that listed individuals will experience is small. This is especially so in comparison to the other anthropogenic and natural stressors that individuals must cope with. Coupled with trends suggesting recovery for most target species, the continuation of close approach activities under the proposed permit are not expected to measurably hamper survival or recovery of listed species.

A possible exception to this is Cook Inlet beluga whales, whose responses to close vessel approach are the most extreme we know of for any cetacean and whose population size is in decline. Exposure is likely to last a short period (average of 2.7 minutes, up to 10 minutes) and individuals have been observed to typically return to baseline behavior very shortly after escaping pursuit. It is apparent that targeted individuals would do everything possible to avoid research vessels at close range and will likely experience significant metabolic costs (exertion likely to anaerobic respiration) and stress responses. Stress levels may rise to the point of distress, with potential muscle damage similar to that documented in other distressed odontocetes (Cowan and

Curry 1998a; Cowan and Curry 2008; Cowan and Curry. 2002). However, to date no severe adverse effects, such as mortality, physiological damage, reduced growth or reproduction have been documented even with several years of monitoring having been conducted. Comparative analysis of stress responses in belugas exposed to similar and more invasive stressors versus captive "baseline" status does not support a distress response. Additional serial study of the stress response to anthropogenic stressors should be undertaken to further clarify the significance of human impacts to the species. At present the available evidence leads us to believe that significant physiological harm can, but is not likely to result from close vessel approaches of Cook Inlet beluga whales and that significant but temporary behavioral and physiological responses would result from approaches to any individual.

In addition to the stressors placed upon targeted individuals from vessel approaches, a portion of the same individuals will be further exposed to stressors associated with biopsy and/or tagging. The *Response analysis* found that responses by whales to these activities are similar to those of vessel approach and are frequently difficult to differentiate (Goodyear 1981; Goodyear 1993b; Hooker et al. 2001c; Mate et al. 1997b; Watkins 1981c; Watkins et al. 1984b). In addition, not all individuals respond to tagging or biopsy, meaning that a fraction of targeted individuals are not expected to show an overt response to the combined approach, tagging, and biopsy action. We do expect all individuals to at least be aware of the vessel's approach and undergo a low-level to high (Cook Inlet beluga whales) stress reaction as a result of a large unknown object in close proximity to individuals. Information available to us does not support behavioral responses by an individual being more severe when additional activities (such as tagging and biopsy) are added to vessel approach, although we do expect more frequent responses to the combined activities versus to approach alone.

Implantable tags also have the potential to introduce pathogenic agents into the blubber and muscle of targeted individuals. This concern has been addressed to some extent by several reviews, but conclusive evidence for or against the potential for infection is lacking (Best and Mate. 2007; Kraus et al. 2000; Mate et al. 2007c; Weller 2008). At present, available evidence from a single animal of advanced decomposition and numerous observations of live whales does not support debilitating infection caused by implantable tags (Weller 2008). Until additional tagging sites of have been evaluated, the issue will likely remain unresolved. Methods adopted by the researchers, including use of disinfectants on biopsy and tagging materials penetrating target individuals, should minimize the risk of infection (Mate et al. 2007c; Weller 2008). We do expect that tagged individuals will exhibit similar inflammatory responses, development of divots, and scar tissue development from implantable tags as has been seen in whale species who have been tagged in the past (Best and Mate. 2007; Kraus et al. 2000; Mate et al. 2007c; Quinn et al. 2000; Weller 2008). We also expect implantably-tagged individuals to experience increased drag during the days to months that tags will be protruding from the blubber as they are rejected from the body (Best and Mate. 2007). Suction cup tags will not likely stay on for as long, but will also cause drag while attached to the target whale. However, we expect the amount of drag to not be significant to target whales, as the tags are small compared to the size of target whales. As evidenced from the apparent ability of whales to survive and reproduce successfully under these conditions (Baumgartner and Hammar. 2010; Best and Mate. 2007; Mate et al. 2007c), we do not expect these physiological responses to be significant to any individuals' overall metabolic balance or health state.

Overall, we expect all targeted whales to experience some degree of stress response to approach, biopsy, and/or tagging attempts. We also expect a fraction of these individuals to undergo short-term behavioral responses to these activities, varying from twitches to evasion and breaching. We do not expect displacement of individuals from the action area as a result of the proposed action. Individuals responding in such ways may temporarily cease feeding, breeding, resting, or otherwise disrupt vital activities. However, we do not expect that these disruptions will cause a measureable impact to any individual's fitness. We expect all biopsied or implantably-tagged individuals to experience additional physiological reactions associated with foreign body penetration into the blubber and possibly muscle, including inflammation, scar tissue development, and (for implantable and suction cup tags) a small amount of drag associated with the applied tags. We do not expect any single individual to experience a fitness consequence as a result of the proposed actions and, by extension, do not expect population-level effects.

Conclusion

After reviewing the current status of endangered blue, fin, sei, humpback, bowhead, North Pacific right, sperm, southern resident killer, Cook Inlet beluga, and Hawaiian insular false killer whales (proposed as endangered), endangered western DPS Steller sea lions, threatened eastern DPS Steller sea lions, Arctic DPS ringed seals (proposed as threatened), and Beringia DPS bearded seals (proposed as threatened) in the *Status of Listed Resources*, the *Environmental Baseline* for the action area, the effects of the proposed research programs, and the *Cumulative Effects*, it is NMFS' biological opinion that issuing Permit 14245 is likely to adversely affect individuals of the forementioned listed species or species proposed for listing but is not likely to jeopardize the continued existence of the species. We do not expect that critical habitat will be adversely affected by the proposed actions.

Incidental Take Statement

Section 9 of the ESA and Federal regulation pursuant to section 4(d) of the ESA prohibits the take of endangered and threatened species, respectively, without special exemption. Take is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct. Harm is further defined by NMFS to include significant habitat modification or degradation that results in death or injury to listed species by significantly impairing essential behavioral patterns, including breeding, feeding, or sheltering. Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. Under the terms of 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 ESA provided that such taking is in compliance with the terms and conditions of this Incidental Take Statement.

We do not expect incidental take of threatened or endangered species as a result of the proposed actions.

Conservation Recommendations

Section 7(a)(1) of the Act directs Federal agencies to utilize their authorities to further the purposes of the Act by carrying out conservation programs for the benefit of endangered and threatened species. Conservation recommendations are discretionary agency activities to

minimize or avoid adverse effects of a proposed action on listed species or critical habitat, to help implement recovery plans, or to develop information.

The following conservation recommendations would provide information for future consultations involving the issuance of marine mammal permits that may affect endangered whales as well as reduce harassment related to research activities:

1. *Determination of take numbers.* We find that the methods used to determine the number of takes for most actions proposed to be authorized by the Permits Division to be several orders of magnitude in excess of those that are reasonably likely to occur. We recommend that the Permits Division adopt methods in determining take numbers that are reasonable in light of the actions proposed by the NMML and the level of exposure which has occurred in the past under similar or identical NMML actions as evidenced by annual reports.

2. *Identify responses by listed individuals to permitted actions.* The Endangered Species Division recommends that annual reports submitted to the Permits Division require detail on the response of listed individuals to permitted activities. Expansive, individual-by-individual detail is not recommended, but a minimum of general comments on individual responses can inform future consultations. Information should include responses to targeted individuals to whom actions are directed, unintentional responses of the conspecifics of target individuals, and unintentional exposure to non-target species. This will greatly aid in analyses of likely impacts of future activities.

In order for NMFS Endangered Species Division to be kept informed of actions minimizing or avoiding adverse effects or benefiting listed species or their habitats, the Permits Division should notify the Endangered Species Division of any conservation recommendations they implement in their final action.

Reinitiation Notice

This concludes formal consultation on NMFS' proposal to issue Permit 14245 to the NMML, pursuant to the provisions of section 10 of the ESA and MMPA. 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 authorized take is exceeded, NMFS Permits, Conservation, and Education Division must immediately request reinitiation of section 7 consultation.

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