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 and Conservation Division
Activities Considered:	Issuance of permit to Kenneth Balcomb (Center for Whale Research [Permit 15569]), John Calambokidis (Cascadia Research Collective [Permit 16111]), Jenny Atkinson (The Whale Museum [Permit 16160]), and Brad Hanson (Northwest Fisheries Science Center [Permit 16163])
Consultation Conducted by:	NOAA's National Marine Fisheries Service-Office of Protected Resources-Endangered Species Act Interagency Cooperation Division
Approved by:	
Date:	JUN 04 2012

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 and Conservation Division (Permits Division), which proposes to authorize close approach, aerial survey, and tagging activities on blue, fin, sei, humpback, North Pacific right (no tagging), sperm, southern resident killer, and Hawaiian insular false killer whales in the North Pacific Ocean. Unintentional harassment of both eastern and western DPSs of Steller sea lions, Guadalupe fur seals, and Hawaiian monk seals is also proposed. The consulting agency for this proposal is the NMFS' Office of Protected Resources – Endangered Species Act Interagency Cooperation Division.

This document represents the NMFS' biological and conference opinion (Opinion) of the effects of the proposed actions on endangered and threatened species, as well as species proposed for listing, 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 species, the most current stock assessment reports, past and current research and population dynamics modeling efforts, monitoring reports from prior research, expert opinion, other information provided by the applicant, and biological opinions involving similar research.

Consultation history

On November 3, 2011, the Permits Division published a notice in the Federal Register soliciting public comment on their intent to issue proposed permits 15569, 16160, and 16163.

On January 9, 2012, NMFS' Endangered Species Act Interagency Cooperation Division received a request for formal consultation from the Permits Division to authorize permits 15569, 16111, 16160, and 16163. Consultation was initiated on this date.

Description of the proposed action

The applicants propose to conduct numerous research projects on marine mammals to aide in ongoing studies of habitat use, population size and structure, social organization, range, movement patterns, health assessment, energetic, movement rates, diving behavior, diet, ecology, and behavior, and impact of human activities. These studies include aerial and vessel-based transect surveys as well as small boat approaches of marine mammals for photoidentification, passive acoustic recording, sound playback experiments, ultrasound, biopsy, breath sampling, and telemetry tagging (both non-invasive suction-cup tagging as well as dermal [dart] tagging). The use of unmanned aerial vehicles (remote controlled helicopters) is also proposed.

Aerial surveys-permits 15569, 16111, and 16163

Aerial surveys may be conducted from a variety of platforms, including single or twin-engine aircraft such as a Partenavia Observer, as well as helicopters and unmanned airships. We assume that photographs and sighting documentation would be recorded from the aircraft as is typically done during aerial surveys. Aerial surveys may also be conducted to assist surface vessels in locating target individuals. In this situation, a survey aircraft would target an individual or group and then circle or otherwise maintain contact until the vessel arrives. Persons onboard the vessel would photoidentify, biopsy sample, and/or deploy telemetry devices. Flight altitude would vary, but could range from 215-450 m, although tethered airships may operate at 60-120 m. Airships would be tethered to a 20-40 foot long research vessel, which may approach listed individuals to orient the airship, depending upon environmental conditions. Over the course of a survey season, an individual or group of marine mammals may be overflown multiple times. Transect surveys may be flown.

Vessel survey-permits 15569, 16111, 16160, and 16163

Vessel approaches would follow marine mammals to obtain photoidentification records (under all proposed permits) or attach a telemetry device to a target individual(s). Vessels of 5-50+ m in length (usually 5-14 m) would approach an individual laterally or from behind using the minimum speed required to close the distance with the target individual or group. Large vessels may deploy 200-400 m long towed passive acoustic arrays. For some species, photographs of both right and left sides would be conducted, entailing at least an approach, a retreat, and a re-approach. Focal follows before or after tagging attempts may be conducted, largely from a distance that would not disturb individuals (50-500 m) and total time in association with a target individual or group may be up to several hours, although approaches for photo-identification generally last from a few minutes to an hour. Approaches would be conducted from the side or rear of a target individual. For some species, such as southern resident killer whales, individuals may be approached up to 100 times annually and ten times per day.

Under permit 15569, more than one research vessel may be deployed simultaneously in the same area. Approaches for photoidentification, breath sampling, behavioral observation, and tagging would close to within 2-30 m. Permit 15569 would allow up to five daily approaches for individual humpbacks. Intentional approaches of snorkelers or divers would be limited to within 5 m of an individual marine mammal and 10 minutes in duration and occur only for blue, fin, or humpback whales. In lieu of in-water personnel, a pole-mounted camera may be used.

Experimental pile driving, white noise, and ocean noise playbacks-permit 16163

Experimental playbacks of control and simulated pile driving sounds would be conducted in inland waters of Washington State targeting southern resident killer whales. The sound source would be omnidirectional and start at 110 dB re: 1 μPa @ 1 m. The source level would be increased in 5 dB increments every 10 s until a response is observed visually or a maximum source level is reached (180 dB re: 1 μPa @ 1 m). Sounds would include recordings of vibratory and impact pile drivers (or simulated versions of these) as experimental treatments as well as control treatments, including sounds of precipitation underwater, wave noise, and white noise. Experiments would include baseline observation as well as observation during and after exposure from a small research vessel. Post-exposure observation would last 30-60 minutes, or until individuals return to baseline behavior, whichever comes last. Playbacks would preferably occur to groups with at least one individual carrying a suction-cup based tag that records received sound. Playbacks would be immediately discontinued if an individual displayed aggression, severe avoidance, or if group cohesion extensively changed for prolonged periods. Groups with calves younger than one year old would be avoided. The applicant aims to expose ten groups annually where an individual in each group carries an acoustic tag. Multiple annual exposures may occur. Exposures to the same individual more than once per day and five times annually would be avoided.

Echosounding prey and whale imaging-permits 16111 and 16163

A 34-462 kHz commercial depth sounder or echosounder would be used to examine prey occurrence. Under permit 16163, marine mammals would be imaged with a multi-beam echosounder using frequencies around 100-240 kHz. Permit 16111 would authorize acoustic recordings undertaken from a vessel using hydrophones. Recording from a vessel would entail the approach of an individual marine mammal or group to within 1,000 m in front of the marine mammal(s), halting, and allowing target individuals to pass by.

Breath sampling-permits 16111 and 16163

Breath sampling would occur by attaching a collection plate or nylon mesh catch system to a 3-6 m aluminum or carbon fiber pole and placed over the target individual (particularly bowriding individuals) as it surfaces to exhale; no physical contact will occur between the target individual and the experimental devices. A vacuum system may be employed. Permit 16163 would authorize the use of remote controlled helicopters or blimps for breath sampling, where culture plates would be mounted on the vehicle hovering over target individuals as they exhale. An individual may be approached up to three times in attempts to collect breath samples.

Biopsy sampling-permits 16111 and 16163

Biopsy samples would be obtained either through collecting sloughed skin found on or near the water's surface after a target individual has passed by, or by projecting a sterilized biopsy dart onto a target individual and obtaining a blubber core. Collecting sloughed skin would not necessarily involve close approach. Biopsy samples would be collected using a crossbow-deployed arrow with a stainless steel biopsy punch 7-9 mm in width and 2-5 cm in depth (depending upon the estimated blubber thickness of the target). A tether would not be frequently used, but may in some occasions. The dorsal fin area would be targeted for sampling. The floating arrow would be collected after bouncing or detaching from a whale. Vessel approaches to 5-30 m would be necessary.

Dart tagging-permits 16111 and 16163

Tagging may occur using several methods (4-7 m long pole, pneumatic rifle, or crossbow) and instrument types (dart, physiological, and suction cup) from 2-30 m away. Tags would be applied to all age classes except calves less than six months old and females accompanied by calves less than six months of age. Relatively accurate dates of birth are known for southern resident killer whales, but the individual's age would be estimated by field researchers based on the time of year (expectation of calving period and juveniles becoming independent) and presence of fetal folds or fluke curvature for other species. Dart tags implant using dermal anchors (blubber implantation only up to 6-7 cm). Depth of penetration into blubber would vary depending upon the tag type and target species. Smaller species would be targeted with shorter barbed tags. However, barbs have the potential to penetrate the blubber-muscle interface if applied outside the target area, but should not if applied per the applicant's methods. All species may receive suction-cup tags, but no individual would be intentionally tagged (or attempted to be tagged) more than three times. Dart tags would be deployed using a crossbow or airgun aimed at the dorsal fin or just below it. Crossbows would involve the use of arrows, which normally bounce free after tag attachment or come off soon after submersion. 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 used 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, as tags would be no larger or otherwise be more impactful than those assessed within the context of this consultation.

Tags proposed for use fall within one of three categories: 1. suction-cup device (up to 2,400 g, but usually 1,200 g and 19.3 cm long by 3.2 cm wide or less) that measures temperature, light, sound, heading, orientation, and/or emits radio signals beyond the range of a target individual's hearing (or that of its predators or prey) for hours to days, and may have a video/still recorder; 2. physiological tag: recoverable instruments with two electrodes (up to 4 cm wide) connected by a 40 cm wire and an attached data logger that connect to the target individual by suction cup (total weight 200-400 g; 24 cm by 8 cm by 8 cm) for hours to 2 days; 3. satellite transmitter: long-term (days to one year), implantable device (5.3-6.3 cm long by 2.2-2.4 cm high by 3.0-5.2 cm wide [44-59 g] for dart tags) that can couple with additional sensing instruments and transmit information to orbiting satellites.

Suction cup tags not only provide telemetry data, but normally include accessory sensing instrumentation. From one to four suction cups may be incorporated. Attachment generally lasts for a few hours to a few days. As with fully implantable tags, the target region is high on the individual's back.

Dart tags are essentially miniaturized satellite transmitters. Tags are fired with crossbows, poles, or airguns, with the tag being deployed with an arrow that is subsequently retrieved (except with pole deployment) at distances of 2-30 m. Attachment is via a pair of titanium-barbed darts that penetrate up to 12 cm, but frequently less based upon expectations of target blubber depth; the goal is to avoid penetrating the blubber-muscle interface. These darts would be 0.6 cm in diameter and have multiple backwards facing petals to retard movement out of the body. Tags are expunged from the body over the course of weeks to as long as a year. The target region is the dorsal fin or just beneath it.

An individual could be targeted to carry up to two tags in a given year (except southern resident killer whales: one) and four attempts at tagging would be authorized per year (three in a day, except for southern resident killer whales: two). Although individual discrimination is not possible for several listed species, catalogs of false killer whales and southern resident killer whales are available to researchers in the field and should ensure that researchers know what exposure individuals have received in the past and ensure that excessive exposure does not occur.

For southern resident killer whales, up to four attempts per year, two attempts per day, and one successful deployment at any given time is proposed for permit 16163.

Permit Conditions

I. Number and kind(s) of protected species, location(s) and manner of taking

1. The tables in appendix 1 of each permit outline the number of protected species, by species and stock, authorized to be taken, and the locations, manner, and time period in which they may be taken.

2. Researchers must immediately stop permitted activities and the permit holder must contact the Chief of the NMFS Permits Division for written permission to resume:

    a. If serious injury or mortality of protected species occurs.

    b. If authorized take is exceeded, including accidental takes of protected species not listed in each permit.

    c. The permit holder must cease dart tagging of southern resident killer whales in the event dart breakage occurs (i.e. dart barbs are separated from the tag sensor package and remain implanted) and notify the Permits Division Chief by telephone (301-427-8401) within two days of the event; and, submit an incident report that includes a complete description of the events surrounding the incident and identification of steps that will be taken to reduce the potential for additional breakage occurrence. Dart tagging southern resident killer whales may recommence upon review of that information and authorization by the Permits Division Chief.

3. Researchers working under each permit may collect visual images (*e.g.*, photographs, video) in addition to the photo-identification or behavioral photo-documentation

authorized in appendix 1 of each permit as needed to document the permitted activities, provided the collection of such images does not result in takes.

4. The permit holder may use visual images and audio recordings collected under his/her permit, including those authorized in appendix 1 of each permit, in printed materials (including commercial or scientific publications) and presentations provided the images and recordings are accompanied by a statement indicating that the activity was conducted pursuant to the particular permit. This statement must accompany the images and recordings in all subsequent uses or sales.

5. U pon written request from the permit holder, approval for photography, filming, or audio recording activities not essential to achieving the objectives of the permitted activities, including allowing personnel not essential to the research (*e.g.*, a documentary film crew) to be present, may be granted by the Permits Division Chief.

   a. The permit holder submits a request to the Permits Division specifying the location and nature of the activity, approximate dates, and number and roles of individuals for which permission is sought.

   b. Non-essential photography, filming, or recording activities will not influence the conduct of permitted activities or result in takes of protected species.

   c. Persons authorized to accompany the researchers for the purpose of such non-essential activities will not be allowed to participate in the permitted activities.

   d. The permit holder and researchers do not require compensation from the individuals in return for allowing them to accompany Researchers.

6. Researchers must comply with the following conditions related to the manner of taking:

   a. <u>Counting and reporting takes</u>

   i. Any "approach" of a cetacean constitutes a take by harassment and must be counted and reported.

   ii. Regardless of success, any attempt, which includes the associated close approach, to sample an animal constitutes a take and must be counted and reported.

   iii. No individual animal may be taken more than 3 times in one day (4 for Permit 16163).

   iv. For MMPA Level A procedures (tag/sample):

a. Each additional attempt to perform the suite of procedures during the same approach constitutes a new take and must be counted and reported against that row of takes.

b. Attempts include misses, successful hits, and hits with no data or sample collected.

c. No more than two tagging attempts by dart or suction cup tagging and three attempts for biopsy and ultrasound sampling per encounter.

d. Any marine mammal observed during sound playback must be counted as a take by harassment and reported.

e. During aerial surveys flown at an altitude lower than 1,000 feet, any cetacean observed should be counted and reported as a take.

b. General

i. To minimize disturbance of the subject animals the permit holder must exercise caution when approaching animals and must retreat from animals if behaviors indicate the approach may be interfering with reproduction, feeding, or other vital functions.

ii. Where females with calves are authorized to be taken, researchers:

a. Must immediately terminate efforts if there is any evidence that the activity may be interfering with pair-bonding or other vital functions;

b. Must not position the research vessel between the mother and calf;

c. Must approach mothers and calves gradually to minimize or avoid any startle response; and

d. Must not approach any mother or calf while the calf is actively nursing.

e. Must, if possible, sample the calf first to minimize the mother's reaction when sampling mother/calf pairs.

c. For underwater filming and/or photography:

i. No more than two divers must be in the water at any time during underwater observations. An underwater approach/activity must be terminated if a whale is observed to exhibit adverse/evasive changes in behavior. Use of an additional diver is subject to review and approval by the Permits Division.

ii. With the exception of professional and/or experienced photographers/videographers, research assistants are not authorized to carry out underwater observations and/or photography.

d. Non-target species

i. These permits do not authorize takes of any protected species not identified in appendix 1 of each permit, respectively, including those species under the jurisdiction of the USFWS. Should other protected species be encountered during the research activities authorized under these permits, researchers must exercise caution and remain a safe distance from the animal(s) to avoid take, including harassment.

e. Aerial surveys

i. Manned and un-manned aerial surveys (excluding tethered airships) must be flown at no less than an altitude of 300 feet. Aerial surveys will be flown at an altitude of 750 feet for most species under permit 16111.

ii. To minimize disturbance: If an animal shows a response to the presence of the aircraft, the aircraft must leave the vicinity and either resume searching or continue on the line-transect survey.

iii. Aerial flights must not be conducted over marine mammal haul out areas.

f. Biopsy, tagging, and ultrasound sampling

i. All biopsy tips must be disinfected between and prior to each use.

ii. Under permit 16163, researchers may biopsy sample and/or tag calves greater than one year old and females accompanied by these calves. However, no calf less than one year old or female accompanied by such a calf shall be sampled. Under permit 16111, researchers may biopsy sample adults, calves greater than four months (blue, fin, humpback whale) or one year (all other species) old, and females accompanied by these calves. However, no calf less than four months or one year old or female accompanied by a calf less than one year old shall be sampled.

iii. Before attempting to sample an individual, researchers must take reasonable measures (e.g., compare photo-identifications) to avoid repeated sampling of any individual.

iv. A tag attachment, ultrasound, or biopsy attempt must be discontinued if an animal exhibits repetitive strong adverse reactions to the activity or the vessel.

v. No individual animal may be tagged with both suction cup and implantable tags more than once per year per year.

vi. In no instance will a permit holder attempt to biopsy or tag a cetacean anywhere forward of/anterior to the pectoral fin under permit 16111.

g. Active acoustics

i. Playback studies must be limited to one trial per day and five annually per individual, not to exceed 179 dB re 1 µPa at 1 meter.

ii. A playback episode must be discontinued if an animal exhibits repetitive strong adverse reactions to the playback activity or the vessel (e.g. aggression, aversion, avoidance, or extensive/prolonged changes in group cohesion).

iii. Researchers must take reasonable measures to avoid playback exposure to groups that include calves less than one year old.

II. Qualifications, responsibilities, and designation of personnel

1. At the discretion of the permit holder, the following researchers may participate in the conduct of the permitted activities in accordance with their qualifications and the limitations specified herein:

   a. Principal investigator – Kenneth C. Balcomb III (Permit 15569), Kari Koski (Permit 16160), M. Bradley Hanson (Permit 16163), and John Calambokidis (Permit 16111)

   b. Co-investigator(s) – Susan Berta, John Durban, David Kay Ellifrit, Holly Fearnbach, Emma Foster, Howard Garrett, Erin Heydenreich, Astrid Maria van Ginneken (Permit 15569), John Calogero and Jeff Hogan (Permit 16160), Candice Emmons, Dawn Noren, Marla Holt, Jeff Foster, Robin Baird, Daniel Webster, John Calmbokidis, Erin Falcone, Greg Schorr, Allan Lignon (Permit 16163), and Robin Baird, Annie Douglas, Erin Falcone, Jeff Foster, Jessie Huggins, Jeff Jacobsen, Lisa Schlender, Greg Schorr, and Gretchen Steiger (Permit 16111)

   b. Research assistants – personnel identified by the permit holder or principal investigator and qualified to act pursuant to conditions C.2, C.3, and C.4 of each permit

2. Individuals conducting permitted activities must possess qualifications commensurate with their roles and responsibilities. The roles and responsibilities of personnel operating under these permits are as follows:

   a. The permit holders are ultimately responsible for activities of individuals operating under the authority of each respective permit.

   b. The principal investigators (PI) are the individual primarily responsible for the taking, import, export and related activities conducted under the permit. The PI must be on site during activities conducted under respective Permits unless a co-investigator named in condition C.1 of a permit is present to act in place of the PI.

   c. Co-investigators (CIs) are individuals who are qualified to conduct activities authorized by the Permit without the on-site supervision of the PI. CIs assume the role and responsibility of the PI in the PI's absence.

   d. Research assistants (RAs) are individuals who work under the direct and on-site supervision of the PI or a CI. RAs cannot conduct permitted activities in the absence of the PI or a CI.

3. Personnel involved in permitted activities must be reasonable in number and essential to conduct of the permitted activities. Essential personnel are limited to:

   a. Individuals who perform a function directly supportive of and necessary to the permitted activity (including operation of vessels or aircraft essential to conduct of the activity);

   b. Individuals included as backup for those personnel essential to the conduct of the permitted activity; and

   c. Individuals included for training purposes.

4. Persons who require state or federal licenses to conduct activities authorized under the permits (*e.g.*, veterinarians, pilots) must be duly licensed when undertaking such activities.

5. Permitted activities may be conducted aboard vessels or aircraft, or in cooperation with individuals or organizations, engaged in commercial activities, provided the commercial activities are not conducted simultaneously with the permitted activities.

6. The permit holders cannot require or receive direct or indirect compensation from a person approved to act as PI, CI, or RA under this permit in return for requesting such approval from the Permits Division

III. Reports

1. The permit holders must submit written annual, final, and incident reports to the Permits Division. Reports may be submitted
   - through the online system at https://apps.nmfs.noaa.gov,
   - by email attachment to the permit analyst for this permit, or
   - by hard copy mailed or faxed to the Chief, Permits Division, Office of Protected Resources, NMFS, 1315 East-West Highway, Suite 13705, Silver Spring, Maryland 20910; phone (301) 427-8401; fax (301) 713-0376.

2. Written incident reports related to serious injury and mortality events or to exceeding authorized takes, must be submitted to the Permits Division Chief within two weeks of the incident. The incident report must include a complete description of the events and identification of steps that will be taken to reduce the potential for additional research-related mortality or exceedence of authorized take. In addition to the written report, permit holders must contact the Permits Division by phone (301-427-8401) as soon as possible, but no later than within two business days of the incident.

3. An annual report must be submitted to the Permits Division Chief at the conclusion of each year for which the permit is valid. The annual report describing activities conducted during the previous permit year must follow the format in appendix 2 of each permit.

4. A final report must be submitted to the Permits Division Chief within 180 days after expiration of each permit, or, if the research concludes prior to permit expiration, within 180 days of completion of the research. The final report must follow the format in appendix 2 of each permit.

5. Research results must be published or otherwise made available to the scientific community in a reasonable period of time. Copies of technical reports, conference abstracts, papers, or publications resulting from permitted research must be submitted the Permits Division.

IV. Notification and coordination

1. Permit holders must provide written notification of planned field work at least two weeks prior to initiation of each field trip/season. If there will be multiple field trips/seasons in a permit year, a single summary notification may be submitted per year.

   a. Notification must include the
      - locations of the intended field study and/or survey routes
      - estimated dates of activities

    - number and roles of participants (for example: PI, CI, veterinarian, boat driver, safety diver, animal restrainer, research assistant "in training")

b. Notification must be sent to the following assistant regional administrator(s) for Protected Resources:

   For activities in Alaska; Arctic Ocean; and Bering, Beaufort, and Chukchi Seas: Alaska Region, NMFS, P.O. Box 21668, Juneau, Alaska 99802-1668; phone (907)586-7235; fax (907)586-7012;

   For activities in Washington and Oregon: Northwest Region, NMFS, 7600 Sand Point Way NE, BIN C15700, Bldg. 1, Seattle, Washington 98115-0700; phone (206)526-6150; fax (206)526-6426;

   For activities in California and Antarctic: Southwest Region, NMFS, 501 West Ocean Blvd., Suite 4200, Long Beach, California 90802-4213; phone (562)980-4020; fax (562)980-4027.

   For activities in Hawaii, American Samoa, Guam, and Northern Mariana Islands: Pacific Islands Region, NMFS, 1601 Kapiolani Blvd., Suite 1110, Honolulu, Hawaii 96814-4700; phone (808)944-2200; fax (808)973-2941.

2. To the maximum extent practical, permit holders must coordinate permitted activities with activities of other permit holders conducting the same or similar activities on the same species, in the same locations, or at the same times of year to avoid unnecessary disturbance of animals. Contact the applicable regional office(s) listed in F.1.b of each permit for information about coordinating with other permit holders.

3. Researchers must comply with protocols provided by the regional administrators related to coordination of research, including additional measures deemed necessary to minimize unnecessary duplication, harassment, or other adverse impacts from multiple permit holders.

V. Observers and inspections

1. NMFS may review activities conducted pursuant to each permit. At the request of NMFS, permit holders must cooperate with any such review by:

   a. Allowing an employee of NOAA or other person designated by the NMFS Office of Protected Resources Director to observe permitted activities; and

   b. Providing all documents or other information relating to the permitted activities.

VI. Modification, suspension, and revocation

1. Permits are subject to suspension, revocation, modification, and denial in accordance with the provisions of subpart D [Permit Sanctions and Denials] of 15 CFR part 904.

2. The NMFS Office of Protected Resources Director may modify, suspend, or revoke a permit in whole or in part:

    a. In order to make the permit consistent with a change made after the date of permit issuance with respect to applicable regulation prescribed under section 103 of the MMPA and section 4 of the ESA;

    b. In a case in which a violation of the terms and conditions of the permit is found;

    c. In response to a written request from the permit holder;

    d. If NMFS determines that the application or other information pertaining to the permitted activities (including, but not limited to, reports pursuant to section E of each permit and information provided to NOAA personnel pursuant to section G of each permit) includes false information; and

    e. If NMFS determines that the authorized activities will operate to the disadvantage of threatened or endangered species or are otherwise no longer consistent with the purposes and policy in section 2 of the ESA.

3. Issuance of these permits does not guarantee or imply that NMFS will issue or approve subsequent permits or amendments for the same or similar activities requested by the permit holders, including those of a continuing nature.

VII. Penalties and permit sanctions

1. A person who violates a provision of this permit, the MMPA, ESA, or the regulations at 50 CFR 216 and 50 CFR 222-226 is subject to civil and criminal penalties, permit sanctions, and forfeiture as authorized under the MMPA, ESA, and 15 CFR part 904.

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 or proposed 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 or proposed 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 or proposed 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 or proposed 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 or proposed 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 or proposed 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 or proposed 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 or proposed 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, endangered, or proposed 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, expert opinion, 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 Bureau of Ocean Energy Management, 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, 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 (southern resident killer and Hawaiian insular false killer whales are possible exceptions) 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 past annual monitoring reports from the applicants and documentation provided by the Permits Division, we assume that proposed activities will be similar to those that the applicant 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 travel over wide areas and although they likely occupy restricted regions for relatively brief periods (hours to days), individuals are 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 the applicants' annual reports and other specific information provided, 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

All proposed permits would occur principally in nearshore marine waters along the U.S. west coast from California to Washington State and, for permits 15569 and 16160, particularly or exclusively in inland marine waters of Washington State. Additional research may be focused in offshore waters of the U.S. west coast (all proposed permits), marine areas along Alaska (proposed permits 15569, 16111, and 16163) and Hawaii (proposed permit 16111 and 16163). Actions may be conducted in the U.S. exclusive economic zone (EEZ) or territorial waters of Canada (all permits). The applicants would be permitted to conduct research during any time of year.

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.

Common name (distinct population segment, evolutionarily significant unit, or subspecies)	*Scientific name*	*Status*
Cetaceans		
Blue whale	*Balaenoptera musculus*	Endangered
Bowhead whale	*Balaena mysticetes*	Endangered
Fin whale	*Balaenoptera physalus*	Endangered
Humpback whale	*Megaptera novaeangliae*	Endangered
Killer whale (Southern Resident*)	*Orcinus orca*	Endangered
North Pacific right whale*	*Eubalaena japonica*	Endangered
Sei whale	*Balaenoptera borealis*	Endangered
Sperm whale	*Physeter macrocephalus*	Endangered
Pinnipeds		
Guadalupe fur seal	*Arctocephalus townsendi*	Threatened
Hawaiian monk seal*	*Monachus schauinslandi*	Endangered
Steller sea lion (Eastern*)	*Eumetopias jubatus*	Threatened
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 imbricate*	Endangered
Kemp's ridley sea turtle	*Lepidochelys kempii*	Endangered
Leatherback sea turtle*	*Dermochelys coriacea*	Endangered
Loggerhead sea turtle (North Pacific)	*Caretta caretta*	Endangered
Olive ridley sea turtle (Mexico's Pacific coast breeding colonies)	*Lepidochelys olivacea*	Endangered

Common name (distinct population segment, evolutionarily significant unit, or subspecies)	***Scientific name***	***Status***
Olive ridley sea turtle (All other areas)		Threatened
Fishes		
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
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
Bearded seal (Beringia)	*Erignathus barbatus nauticus*	Proposed Threatened
Ringed seal (Arctic)	*Phoca hispida hispida*	Proposed Threatened

Species not considered further

Guadalupe fur seals have been documented as far north as Washington State, but primarily in the Farallon Islands and offshore islands of southern California (Belcher and T.E. Lee 2002; Carretta et al. 2002; Reeves et al. 2002b). Occurrence in these regions is extralimital or rare and we do not expect individuals to co-occur with the proposed action in space and time. If co-occurrence does occur, we do not expect aerial or vessel activities to adversely affect any individual. Research on Hawaiian monk seals has routinely involved aircraft overflights; individuals seem to be generally oblivious to overflights, with only occasional head raises observed in response to large low-flying aircraft (NMFS 2009a). We do not expect Hawaiian monk seals to respond to small, higher flying survey aircraft. We also do not expect Hawaiian monk seals to be exposed to vessel approaches. Most individuals occur in the Northwestern Hawaiian Islands, where researcher presence is uncommon. Although some individuals do occur in the main Hawaiian Islands, the probability of an individual being exposed at a meaningful range to proposed activities from a research vessel are discountably low.

We also cannot identify any aspect of the proposed action that would adversely impact designated critical habitat for Hawaiian monk seals. The marine component of this habitat was designated primarily as feeding areas for Hawaiian monk seals, while terrestrial habitat serves as pupping and nursing habitat for mothers and pups, although no primary constituent elements were identified with the listing. The applicants would not undertake aerial surveys in areas of Steller sea lion critical habitat or rookeries.

Although one applicant may undertake activities in Alaskan waters, co-occurrence with Beringia DPS bearded and Arctic DPS ringed seals as well as bowhead whales is not expected due to the expected occurrence of these taxa in other locations. Therefore, these species are not considered further in this Opinion.

Listed sea turtles, salmonids, rockfish, eulachon, and southern DPS green sturgeon may also be exposed to potential stressors from the proposed actions. Sea turtles have not been documented to be struck by researchers in the area. 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 sea turtles, salmonids, rockfish, and sturgeon to be discountable. Sounds associated with playback and prey mapping activities should not be audible to these species. For these reasons, we will not consider sea turtles, eulachon, or sturgeon further in this Opinion. We do consider salmonids and rockfishes of the Puget Sound region due to their potential exposure and response to simulated pile driving activities.

We do not expect any aspect of the action to adversely affect green sturgeon, eulachon, or proposed leatherback sea turtle critical habitat. For green sturgeon, primary constituent elements for critical habitat designated in the marine environment include food resources, water quality, and migratory corridors. Eulachon critical habitat in marine waters includes areas of nearshore and offshore marine foraging habitat with water quality and available prey, supporting juveniles and adult survival. The primary constituent elements for proposed leatherback sea turtle critical habitat in the action area include (1.) the occurrence of prey species, primarily scyphomedusae of the order Semaeostomeae (*Chrysaora*, *Aurelia*, *Phacellophora*, and *Cyanea*) of sufficient condition, distribution, diversity, and abundance to support individual as well as population growth, reproduction, and development and (2.) migratory pathway conditions to allow for safe and timely passage and access to/from/within high use foraging areas. The stressors associated with the proposed action, such as aerial and surface transit, would produce low levels of pollution through gasoline combustion. We do not expect these levels to be high enough to significantly impact critical habitat. Local noise fields, such as from boat and aircraft engines, playback of simulated pile driving sounds, white noise, and ocean noise, and fish- and whale-finding sonars, are also not expected to impact any aspect of critical habitat.

Although listed invertebrates (black and white abalone) would co-occur with the proposed actions, we cannot identify any stressors that reasonably could impact their biology, nor adversely impact the proposed critical habitat of black abalone. The primary constituent elements associated with the proposed critical habitat include (1.) rocky benches formed from consolidated rock of various geological origins (e.g., igneous, metamorphic, and sedimentary) that contain channels with macro- and micro-crevices or large boulders (greater than or equal to 1 m in diameter) and occur from mean higher high water to a depth of 6 m, (2.) bacterial and diatom films, crustose coralline algae, and a source of detrital macroalgae, (3.) rocky intertidal habitat containing crustose coralline algae and crevices or cryptic biogenic structures (e.g., urchins, mussels, chiton holes, conspecifics, anemones), (4.) Suitable water quality includes temperature (i.e., tolerance range: 12 to 25 °C, optimal range: 18 to 22 °C), salinity (i.e., 30 to 35 ppt), pH (i.e., 7.5 to 8.5), and other chemical characteristics necessary for normal settlement, growth, behavior, and viability of black abalone, and (5.) Suitable circulation patterns are those that retain eggs, sperm, fertilized eggs and ready-to-settle larvae within 100 km from shore. The proposed action's stressors, such as aerial and surface transit, would produce low levels of pollution through gasoline combustion. We do not expect these levels to be high enough to significantly impact critical habitat. Local noise fields, such as from boat and aircraft engines, playback of simulated pile driving sounds, white noise, and ocean noise, and fish- and whale-finding sonars, are also not expected to impact any aspect of critical habitat.

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.

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. 1998). For management purposes, the International Whaling Commission (IWC) considers all Pacific blue whales to be a single stock, whereas under the MMPA, the NMFS recognizes four stocks of blue whales: western North Pacific Ocean, eastern North Pacific Ocean, Northern Indian Ocean, and Southern Hemisphere.

Until recently, blue whale population structure had not been tested using molecular or nuclear genetic analyses (Reeves et al. 1998). A recent study by Conway (2005) suggested that the global population could be divided into four major subdivisions, which roughly correspond to major ocean basins: eastern North and tropical Pacific Ocean, Southern Indian Ocean, Southern Ocean, and western North Atlantic Ocean. 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 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 1985a). Yochem and Leatherwood (1985a) 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;

"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.

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. 1996b; 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 1985a). 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 2003b; Stafford et al. 2001; Watkins et al. 2000c), 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).

Blue whales from both the eastern and western North Pacific have been heard, tracked, or harvested in waters off Kodiak Island; acoustic detections are made in the Gulf of Alaska from mid-July to mid-December and a peak from August through November (COSEWIC 2002b; Ivashin and Rovnin. 1967; Moore et al. 2006; Stafford 2003a; Stafford et al. 2007; Yochem and Leatherwood 1985b). Although acoustic detections in the Gulf of Alaska were absent since the late 1960s, recordings have increased during 1999-2002 and a few sightings have been made in the northern Gulf of Alaska (Calambokidis et al. 2009b; Moore et al. 2006; NOAA 2004; Stafford 2003a; Stafford et al. 2007; Stafford and Moore 2005). However, surveys in the western Gulf of Alaska and east of Kodiak Island have not found blue whales (Rone et al. 2010; Zerbini et al. 2006). Blue whales are rarely observed in nearshore Alaskan waters, but seem to prefer continental shelf edge waters; such areas in the Gulf of Alaska were formerly feeding grounds for blue whales prior to severe depletion (Rice and Wolman. 1982). Call detections of blue whales from the western North Pacific indicate a greater likelihood of these individual occurring southwest of Kodiak Island (Stafford 2003a).

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 February-May as well as in November (Gedamke and Robinson. 2010; Sirovic et al. 2009b). Gedamke and Robinson (2010) found blue whales to be particularly numerous and/or vocal north of Prydz Bay, Antarctica based upon sonobuoy deployments. Pygmy blue whales were also frequently heard in Antarctic waters, further south than they had previously been documented (Gedamke and Robinson 2010). Other than a single vocal record in Atlantic waters off Angola, pygmy blue whales have been exclusively documented in the Indian Ocean or western Pacific (Cerchio et al. 2010a; Mccauley and Jenner 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 two populations; the true and pygmy blue whales of the Southern Hemisphere (Gilpatrick and Perryman. 2009), although, recent analyses of vocalizations and photos have linked blue whales found in the Costa Rica Dome to the North Pacific population (Chandler and Calambokidis 2004).

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

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 2002a; NMFS 1998b; Yochem and Leatherwood 1985a). 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 1985a). 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 1985a).

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 1985a). 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} at 1 m in the 17-30 Hz range and pygmy blue whale calls at 175± 1 dB re 1 μPa_{rms} at 1 m in the 17-50 Hz range.

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. 1995b).

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 2002a; Yochem and Leatherwood 1985a); 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). Punt (2010) estimated the rate of increase for blue whales in the central North Atlantic to be 9% annually (3.83 SE) between 1987 and 2001.

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 2002a)
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 1985a)
		300,000	--	--	--	(COSEWIC 2002a)
		--	--	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).

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 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 abundance (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. Punt (2010) estimated the rate of increase for blue whales in the eastern North Pacific to be 3.2% annually (1.4 SE) between 1991 and 2005.

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.

Southern Hemisphere. Estimates of 4-5% for an average rate of population growth have been proposed (Yochem and Leatherwood 1985). However, a recent estimate of population growth for Antarctic blue whales throughout the region was 7.3% (Branch et al. 2007). Punt (2010) estimated the rate of increase for blue whales in the Southern Hemisphere to be 8.2% annually (3.37 SE) between 1978 and 2004. Branch et al. (2007) also included an estimate of 1,700 individuals south of 60°. Antarctic blue whales remain severely depleted with the 1996 estimate only 0.7% of pre-whaling levels (IWC 2005). Blue whales along Chile have been estimated to number between 7and 9% of historical abundance (Williams et al. 2011).

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.

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). Contaminant transfer between mother and calf occurs, meaning that young often start life with concentrations of contaminants equal to their mothers, before accumulating additional contaminant loads during life and passing higher loads to the next generation (Gauthier et al. 1997a; Metcalfe et al. 2004).

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 Southern Hemisphere migrate seasonally from Antarctic feeding areas in the summer to low-latitude breeding and calving grounds in winter. Fin whales tend to avoid tropical and pack-ice waters, with the high-latitude limit of their range set by ice and the lower-latitude limit by warm water of approximately 15° C (Sergeant 1977). Fin whale concentrations generally form along frontal boundary, or mixing zones between coastal and oceanic waters, which corresponds roughly to the 200 m isobath (the continental shelf edge (Cotte et al. 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 1998). In areas north of Cape Hatteras, fin whales account for about 46% of the large whales observed in 1978-1982 surveys (CETAP 1982). Little is known about the winter habitat of fin whales, but in the western North Atlantic, the species has been found from Newfoundland south to the Gulf of Mexico and Greater Antilles, and in the eastern North Atlantic their 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). The singing location of fin whales in the Davis Strait and Greenland has been correlated with sea ice fronts; climate change may impact fin whale distribution and movement by altering sea ice conditions (Simon et al. 2010). A general fall migration 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), although movement of individuals between the western Mediterranean and northeastern Atlantic appears to be common (Bentaleb et al. 2011). However, some fin whales from the northeastern North Atlantic have been tracked into the Mediterranean during winter and overlap in time and space with the Mediterranean population may exist (Castellote et al. 2010). Individuals also tend to associate with colder, saltier water, where steep changes in temperature, and where higher northern krill densities would be expected (Cotte et al. 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. 1999a). 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. 1999a). However, fin whales have been sighted as far as 60° N throughout winter (Mizroch et al. 1999b). 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 are seen during the summer off Oregon, and in summer and fall in the Gulf of Alaska and southeastern Bering Sea (Moore et al. 2000; 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).

Fin whale observations have occurred near Kodiak Island throughout the year, but have been most frequent from April through September as supported by acoustic recording data (Baraff et al. 2005; Moore et al. 2006; Stafford et al. 2009; Stafford et al. 2007; Watkins et al. 2000a; Watkins et al. 2000b; Watkins et al. 2000c; Wynne and Witteveen 2005). However, calls peak in the central North Pacific, Gulf of Alaska, and Aleutian Islands during fall and winter (Moore et al. 1998; Moore et al. 2006; Stafford et al. 2009; Watkins et al. 2000a; Watkins et al. 2000b; Watkins et al. 2000c). Fin whales around Kodiak Island appear particularly concentrated relative to other areas (Baraff et al. 2005; Waite 2003; Wynne and Witteveen 2005; Zerbini et al. 2006). This area constitutes a significant foraging area for fin whales. Group sizes of 12-18 individuals are common (Wynne and Witteveen 2005), although average group sizes of 2.1-2.9 have been found for the Aleutian Islands, Bering Sea, and western Gulf of Alaska (Moore et al. 2002; Wade et al. 2003; Waite 2003).

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

Age distribution. Aguilar and Lockyer (1987) suggested annual natural mortality rates in northeast Atlantic fin whales may range from 0.04 to 0.06. 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). This was recently contradicted by acoustic surveys in the Davis Strait and off Greenland, where singing by fin whales peaked in November through December; the authors suggested that mating may occur prior to southbound migration (Simon et al. 2010). Although seasonal migration occurs between presumed foraging and breeding locations, fin whales have been acoustically detected throughout the North Atlantic Ocean and Mediterranean Sea year-round, implying that not all individuals follow a set migratory pattern (Notarbartolo-Di-Sciara et al. 1999; Simon et al. 2010).

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 current boundaries (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 through 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 (Hain et al. 1992). Individuals or groups of less than five individuals represented about 90% of observations. Barlow (2003) reported mean group sizes of 1.1–4.0 during surveys off California, Oregon, and Washington.

Vocalization and hearing. Fin whales produce a variety of low-frequency sounds in the 10-200 Hz range (Edds 1988; Thompson et al. 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. (1995a) reported the most common sound as a 1 s vocalization of about 20 Hz, occurring in short series during spring, summer, and fall, and in repeated stereotyped patterns during winter. Au (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 (Clark and Ellison. 2004; Erbe 2002b). The source depth of calling fin whales has been reported to be about 50 m (Watkins et al. 1987).

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

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. 1995b).

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 1989a; Cherfas 1989b).

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). The abundance of fin whales in the Baffin Bay-Davis Strait summer feeding area is believed to be increasing (Heide-Jorgensen et al. 2010).

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). Punt (2010) estimated the rate of increase for fin whales in the eastern North Pacific to be 4.8% annually (3.24 SE) between 1987 and 2003.

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.

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 2008c)
	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 1992)
	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°S	--	--	1,735	514-2,956	(IWC 1996)
	South of 30°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).

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

Anthropogenic threats. Fin whales have undergone significant exploitation, but are currently protected under the IWC. Fin whales are still hunted in subsistence fisheries off West Greenland. In 2003, two males and four females were landed and two others were struck and lost (IWC 2005). In 2004, five males and six females were killed, and two other fin whales were struck and lost. 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 (2006c). 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). 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. 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. 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 fin whale 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.

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 (Cerchio et al. 2010b; Reeves et al. 2002a; 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). 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. Humpback whales were recently found to migrate to the northwestern Hawaiian Islands, where singing has been recorded; this may represent an as yet undescribed breeding group, or expansion of breeding from the main Hawaiian Islands (Lammers et al. 2011). 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, Guam, Rota, and Saipan from January-March (Darling and Mori 1993; Eldredge 1991; Eldredge 2003; Rice 1998). 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 2007; Calambokidis 1997; Calambokidis et al. 2001b).

Humpback whales from both the eastern, western, and central North Pacific are known to congregate to feed in waters off Kodiak and the Shumagin Islands (Calambokidis et al. 2009a; Calambokidis et al. 2001a; Urban et al. 2000; Waite et al. 1999; Witteveen et al. 2004). The species is found in the Gulf of Alaska year-round, but are most abundant during summer (beginning in April and peaking in late-August to early September) when foraging opportunities draw many individuals into the region (Baker et al. 1985; Consiglieri et al. 1982; Dahlheim et al. 2008; Stafford et al. 2007; Straley 1990; Waite et al. 1999). During this time, feeding aggregations are found throughout the Kodiak Archipelago, although pelagic areas may also be important foraging areas (Baraff et al. 2005; MMC 2002). Humpback whales have been found in particularly high numbers around Kodiak Island in recent surveys, with a density of 54 individuals per 1,000 km^2 (Waite 2003; Zerbini et al. 2006). Sightings were most frequent during fall off Kodiak (Wynne and Witteveen 2005).

Arabian Sea. 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. 2010).

Southern Hemisphere. Eight proposed stocks, or populations, of humpback whales occur in waters off Antarctica (Figure 1). Individuals from these stocks winter and breed in separate areas and are known to return to the same areas. However, the degree (if any) of gene flow (i.e., adult individuals wintering in different breeding locations) is uncertain (Carvalho et al. 2011). Individuals from breeding grounds in Ecuador are somewhat heterogeneous from individuals in other breeding areas, but appear to maintain a genetic linkage (Felix et al. 2009). Based 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 50° W (Dalla Rosa et al. 2008). 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). Females appear in this area in large numbers well before their male counterparts, frequently accompanied by calves (Barendse et al. 2010). Female movement between breeding locations across years has been documented, bringing into question the genetic discreteness of at least Southern Hemisphere populations (Stevick et al. 2011). However, mixing between some populations has not been found (such as between B2 and C1 groups). Sao Tome appears to be primarily a resting, nursing, and calving area with very little breeding occurring (Carvalho et al. 2011).

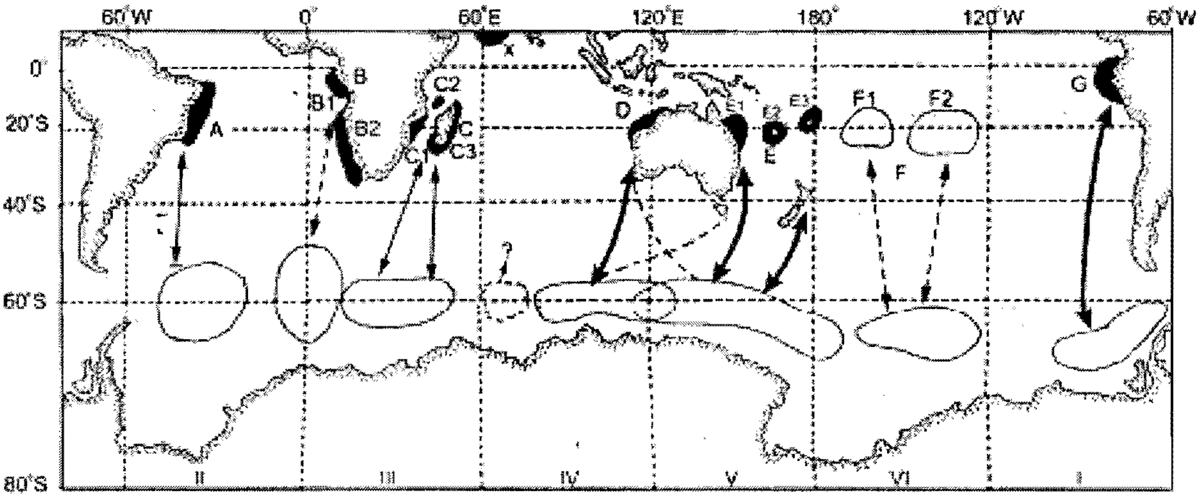

Figure 1. Southern Hemisphere humpback stocks (populations)(IWC 2005).

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 (e.g., southeast Alaska, Gabriele et al. 2007). Females usually breed every 2-3 years, although consecutive calving is not unheard of (Clapham and Mayo 1987; 1990; Glockner-Ferrari and Ferrari 1985 as cited in NMFS 2005b; Weinrich et al. 1993). Males appear to return to breeding grounds more frequently than do females (Herman et al. 2011). Larger females tend to produce larger calves that may have a greater chance of survival (Pack et al. 2009). In some Atlantic areas, females tend to prefer shallow nearshore waters for calving and rearing, even when these areas are extensively trafficked by humans (Picanco et al. 2009).

In calving areas, males sing long complex songs directed towards females, other males, or both. The breeding season can best be described as a floating lek or male dominance polygamy (Clapham 1996). Calving occurs in the shallow coastal waters of continental shelves and oceanic islands worldwide (Perry et al. 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 water depths of less than 182 m. Maximum diving depths are approximately 170 m (but usually <60 m), with a very deep dive (240 m) recorded off Bermuda (Hamilton et al. 1997). Dives can last for up to 21 min, although feeding dives ranged from 2.1-5.1 min in the North Atlantic (Dolphin 1987). In southeast Alaska, average dive times were 2.8 min for feeding whales, 3.0 min for non-feeding whales, and 4.3 min for resting whales (Dolphin 1987). In the Gulf of California, humpback whale dive durations averaged 3.5 min (Strong 1990). Because most humpback prey is likely found within 300 m of the surface, most humpback dives are probably relatively shallow. In Alaska, capelin are the primary prey of humpback and are found primarily between 92 and 120 m; depths to which humpbacks apparently dive for foraging (Witteveen et al. 2008).

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; Witteveen et al. 2011). The principal fish prey in the western North Atlantic are sand lance, herring, and capelin (Kenney et al. 1985). There is good evidence of some territoriality on feeding and calving areas (Clapham 1994; Clapham 1996; Tyack 1981). Humpback whales are generally believed to fast while migrating and on breeding grounds, but some individuals apparently feed while in low-latitude waters normally believed to be used exclusively for reproduction and calf-rearing (Danilewicz et al. 2009; Pinto De Sa Alves et al. 2009). Some individuals, such as juveniles, may not undertake migrations at all (Findlay and Best. 1995). Additional evidence, such as songs sung in northern latitudes during winter, provide additional support to plastic seasonal distribution (Smith and G.Pike 2009). Relatively high rates of resighting in foraging sites in Greenland suggest whales return to the same areas year after year (Kragh Boye et al. 2010).

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

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

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 4 provides estimates of historic and current abundance for ocean regions.

North Atlantic. The best available estimate of North Atlantic abundance comes from 1992-1993 mark-recapture data, which generated an estimate of 11,570 humpback whales (Stevick et al. 2003a). Historical estimates have ranged from 40,000-250,000 (Smith and G.Pike 2009). Smith and Reeves (2010) estimated that roughly 31,000 individuals were removed from the North Atlantic due to whaling since the 1600s. Estimates of animals on Caribbean breeding grounds exceed 2,000 individuals (Balcomb III and Nichols 1982). Several researchers report an increasing trend in abundance for the North Atlantic population, which is supported by increased sightings within the Gulf of Maine feeding aggregation (Barlow 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). Punt (2010) estimated the rate of increase for humpback whales in the Gulf of Maine to be 6.3% annually (1.2 SE). Pike et al. (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 suggest that humpback whales in the area had probably recovered from whaling. However, recent data suggest that the upward growth may have slowed or ceased around Iceland according to analysis of survey data there (Pike et al. 2010).

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 2005). 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 (Barlow et al. (2009) provided a bias-corrected estimate of 20,800 individuals) and the population was growing at 4.9% annually. Almost half of these whales likely occur in wintering areas around the Hawaiian Islands. Punt (2010) estimated the rate of increase for humpback whales in the eastern North Pacific to be 6.4% annually (0.9 SE) between 1992 and 2003 and 10.0% for Hawaii (3.32 SE). Barlow et al. (2011) estimates humpback whales in the North Pacific to number at least 21,063 (CV=0.04) and likely more due to negative biases associated with available data.

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

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

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

Arabian Sea. The population inhabiting the Arabian Sea likely numbers a few hundred individuals at most (Minton et al. 2008). This population likely was much larger prior to exploitation in 1966 by Soviet whaling, with individuals found along not only Oman, but Yemen, Iran, Pakistan, and India (Mikhalev 2000; Minton et al. 2008; Reeves et al. 1991; Slijper et al. 1964; Wray and Martin. 1983).

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). Modeling efforts to bound the number of individuals within Oceania have estimated 2,300-3,500 individuals divided amongst various populations/subpopulations (Constantine et al. 2010). A 2009 spike in calf mortality along western Australia brings into question whether carrying capacity has been reached by this population or other factors have increased mortality (Coughran and Gales 2010). Some vital rates of the humpback whale population summering off eastern Australia (E1) were recently estimated, including adult annual survival of 0.925, subadult survival of 0.70 (Hoffman et al. 2010). Growth rates for certain age classes included 10.7% for adult females and 12.4% for juveniles (Hoffman et al. 2010). Punt (2010) estimated the rate of increase for humpback whales off eastern and western Australia to be 10.9 and 10.1% annually, respectively (0.23 and 4.69 SE, respectively).

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. One-quarter of humpback whales of the Arabian Sea population show signs of tattoo skin disease, which may reduce the fitness of afflicted individuals (Baldwin et al. 2010).

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. Between 30 and 40% of humpback whales in the Arabian Sea show scarring from entanglements, with fishing effort on the rise (Baldwin et al. 2010). Entanglement is estimated to occur to roughly 0.5% of humpback whales breeding off Ecuador annually, with calves being the most affected age class (Alava et al. 2011).

More humpback whales are killed in collisions with ships than any other whale species except fin whales (Jensen and Silber 2003a). 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%. The first estimate of population-level effects of entanglement were recently produced, with over 12% of the Gulf of Maine population of humpbacks acquiring new scars from entanglement interactions annually (Mattila and Rowles 2010).

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

Critical habitat. The NMFS has not designated critical habitat for humpback 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. 2004).

Movement and habitat. Southern Residents are highly mobile and can travel up to 100 miles per day (Baird 2000; Erickson 1978a). 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 2008e).

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 2008e). 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 2008e).

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 2008e). 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 2008e). 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 (Krahn et al. 2009a). 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. As southern resident killer whales are normally exposed to high levels of whale watching, engine exhaust has been assessed as a possible threat and, in some cases, may contribute to health affects (Lachmuth et al. 2011).

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 2008e; 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 2002a; 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 2008e; 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.

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 1935b). 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 1935b). 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 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. 2003a). 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, in 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). North Pacific right whales have also been observed feeding on coccolithophore blooms (Tynan et al. 2001). 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. 2004b). 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). The area where North Pacific right whales are densest in the Gulf of Alaska is between 150 and 170° W and south to 52° N (Shelden and Clapham 2006). A right whale was sighted southeast of Kodiak Island in July 1998 and acoustic detections have been made off Kodiak Island, although no detections occurred from April to August 2003 or in April 2009 (Munger et al. 2008; Rone et al. 2010; Waite et al. 2003b). The greatest frequency of call occurrence in the southerastern Bering Sea occurs from July to October (Munger et al. 2008).

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 40° 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). The current population size of right whales in the North Pacific is likely fewer than 1,000 animals compared to possibly 11,000 individuals or more prior to exploitation (NMFS 1991; NMFS 2006e). Wade et al. (2011) used mark-recapture and genotyping methodologies that produced estimates of 31 and 28 individuals, respectively, for individuals in the Bering Sea (likely representing all individuals from the eastern North Pacific).

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) and Brownell et al. (2001) 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 (Wade et al. (2010) estimated 25-38 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 most individuals are 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 high risk of extinction. Demographic stressors 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. 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 this prey of North Pacific right whales. 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.

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. 2004). The species appears to lack a well-defined social structure and individuals are usually found alone or in small groups of up to six whales (Perry et al. 1999). When on feeding grounds, larger groupings have been observed (Gambell 1985b).

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

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

Historically, sei whales were common in the northern Gulf of Alaska (Calkins 1986b; Consiglieri et al. 1982; Fiscus and Braham 1976; Masaki 1976), but no individuals have been sighted during recent surveys (Wade et al. 2003; Waite 2003; Zerbini et al. 2006; Rone et al. 2009).

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 30° S in the Southern Hemisphere (Reeves et al. 1999). However, confirmed sighting records exist for Papua New Guinea and New Caledonia, with unconfirmed sightings in the Cook Islands (Programme) 2007).

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

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 (Konishi et al. 2009; 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 1986a). 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). Sei whales in the Southern Hemisphere may 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). Sei whales have the flexibility to skim or engulf prey (Brodie and Vikingsson 2009).

Vocalization and hearing. Data on sei whale vocal behavior is limited, but includes records off the Antarctic Peninsula of broadband sounds in the 100-600 Hz range with 1.5 s duration and tonal and upsweep calls in the 200-600 Hz range of 1-3 s durations (McDonald et al. 2005). 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 s, separated by 0.4-

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

Status and trends. The sei whale was originally listed as endangered in 1970 (35 FR 18319), and this status remained since the inception of the ESA in 1973. Table 5 provides estimates of historic and current abundance for ocean regions.

Table 5. 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 2008c)
	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 individuals (Mitchell and Chapman 1977). In 1974, the North Atlantic stock was estimated to number about 2,078 individuals, including 965 whales in the Labrador Sea group and 870 whales in the Nova Scotia group (Mitchell and Chapman 1977). In the northwest Atlantic, Mitchell and Chapman (1977) estimated the Nova Scotia stock to contain 1,393-2,248 whales; an aerial survey program conducted from 1978 to 1982 on the continental shelf and edge between Cape Hatteras, North Carolina, and Nova Scotia generated an estimate of 280 sei whales (CETAP 1982). These two estimates are more than 20 years out of date and likely do not reflect the current true abundance; in addition, the CETAP estimate has a high degree of uncertainty and is considered statistically unreliable (Perry et al. 1999; Waring et al. 2004; Waring et al. 1999). The total number of sei whales in the U.S. Atlantic EEZ remains unknown (Waring et al. 2006). Rice (1977) estimated total annual mortality for adult females as 0.088 and adult males as 0.103.

North Pacific. Ohsumi and Fukuda (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). Between 1991-2001, during aerial surveys, 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. 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 during 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. 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.

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 70° N in the North Atlantic and 70° 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 50° 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. 2004). Genetic studies indicate that movements of both sexes through expanses of ocean basins are common, and that males, but not females, often breed in different ocean basins than the ones in which they were born (Whitehead 2003). Sperm whale populations appear to be structured socially, at the level of the clan, rather than geographically (Whitehead 2003; Whitehead et al. 2008).

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 off 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 through mid-June and from the end of August through mid-November (Rice 1974). They are seen in every season except winter (December-February) off 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). Sperm whales occupying the California Current region are genetically distinct from those in the eastern tropical Pacific and Hawaiian waters (Mesnick et al. 2011). The discreteness of the later two areas remains uncertain (Mesnick et al. 2011).

In the Gulf of Alaska, sperm whales have been sighted along the Aleutian Trench as well as over deeper waters and have been detected acoustically throughout the year (Forney and Brownell Jr. 1996a; Mellinger et al. 2004a). Occurrence is higher from July through September than January through March (Mellinger et al. 2004a; Moore et al. 2006). The vast majority of individuals in the region are likely male based upon whaling records and genetic studies; the area is a summer foraging area for these individuals (Allen and Angliss 2010; Reeves et al. 1985; Straley and O'Connell 2005; Straley et al. 2005). Mean group size has been reported to be 1.2 individuals (Wade et al. 2003; Waite 2003).

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 50° and decrease in proportion to males south of 46-47°.

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

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. 2000a;

Davis et al. 2000b; Davis et al. 2000c; Davis et al. 2002; Wormuth et al. 2000). Surface waters with sharp horizontal thermal gradients, such as along the Gulf Stream in the Atlantic, may also be temporary feeding areas for sperm whales (Griffin 1999; Jaquet et al. 1996; Waring et al. 1993). Sperm whales over George's Bank were associated with surface temperatures of 23.2-24.9° C (Waring et al. 2003).

Local information is inconsistent regarding some aspects of sperm whale habitat utilization. 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.

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

Sperm whale age distribution is unknown, but sperm whales are believed to live at least 60 years (Rice 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). Groups may be stable for long periods, such as for 80 days in the Gulf of California (Jaquet and Gendron 2009). Males start leaving these family groups at about 6 years of age, after which they live in "bachelor schools," but this may occur more than a decade later (Pinela et al. 2009). The cohesion among males within a bachelor school declines with age. During their breeding prime and old age, male sperm whales are essentially solitary (Christal and Whitehead 1997).

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. 2000). Sperm whales typically travel ~3 km horizontally and 0.5 km vertically during a foraging dive (Whitehead 2003). Differences in night and day diving patterns are not known for this species, but, like most diving air-breathers for which there are data (rorquals, fur seals, and chinstrap penguins), sperm whales probably make relatively shallow dives at night when prey are closer to the surface.

Feeding. Sperm whales appear to feed regularly throughout the year (NMFS 2006d). 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). 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 6 contains historic and current estimates of sperm whales by region. Sperm whale populations probably are undergoing the dynamics of small population sizes, which is a threat in and of itself. In particular, the loss of sperm whales to directed Soviet whaling likely inhibits recovery due to the loss of adult females and their calves, leaving sizeable gaps in demographic and age structuring (Whitehead 2003).

Table 6. Summary of past and present sperm whale abundance.

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

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 fisheries operated during the late 1700s to the early 1900s (NMFS 2006d; Townsend 1935a).

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).

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).

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 of strandings remain unclear. Calcivirus and papillomavirus are known pathogens of this species (Lambertsen et al. 1987;

Smith and Latham 1978).

Anthropogenic threats. Sperm whales historically faced severe depletion from commercial whaling operations. From 1800 to 1900, the IWC estimated that nearly 250,000 sperm whales were killed by whalers, with another 700,000 from 1910 to 1982 (IWC Statistics 1959-1983). However, other estimates have included 436,000 individuals killed between 1800-1987 (Carretta et al. 2005b). However, all of these estimates are likely underestimates due to illegal and inaccurate killings by Soviet whaling fleets between 1947-1973. In the Southern Hemisphere, these whalers killed an estimated 100,000 whales that they did not report to the IWC (Yablokov et al. 1998), 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). An individual was caught and released from gillnetting, although injured, on Georges Bank during 1990. A second individual was freed, but injured, from gillnetting on George's Bank in 1995. In 1994, a sperm whale was disentangled from gillnet along the coast of Maine. In August 1993, a dead sperm whale, with longline gear wound tightly around the jaw, was found floating ~32 km off Maine. 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).

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.

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

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

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 (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 lions includes animals west of Cape Suckling, Alaska (144°W; 62 FR 24345). However, individuals move between rookeries and haul out sites regularly, even over long distances between eastern and western DPS locations (Calkins and Pitcher 1982a; Raum-Suryan et al. 2002; Raum-Suryan et al. 2004). Most adult Steller sea lions occupy rookeries during the summer 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 1998). 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.

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 1982b; 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).

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 37 km 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 1982b; Gentry 1970; Pitcher and Calkins 1981; Sandegren 1970; Trites et al. 2006). Pups may enter the water after 2-4 weeks (Sandegren 1970).

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 1982b; NMFS 1992; Pike and Maxwell 1958; Trites and Larkin 1992; York 1994).

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 1986a; Calkins and Pitcher 1982b; Loughlin 1997). Adult females may travel far out to sea into water greater than 1,000 m deep (Merrick and Loughlin 1997). Studies on immature Steller sea lions indicate three types of movements: long-range trips (greater than 15 km and greater than 20 hours), short-range trips (less than 15 km and less than 20 hours), and transits to other sites (NMFS 2007). 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 480 km 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). Large numbers of Steller sea lions are found near the 200 m isobath year round (Consiglieri et al. 1982). Foraging generally occurs within 8-24 km of shore (Fiscus and Braham 1976). However, foraging can occur hundreds of kilometers from shore over extended periods (Merrick et al. 1997).

Six major rookeries and numerous haulouts occur in the action area, some of which are in decline (Sease and Gudmundson 2002; Wynne et al. 2005).

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). Adult females embark on foraging trips of at night for 7-26 hours during the breeding season, while adult males rarely or never eat while on breeding grounds (Andrews et al. 2001; Loughlin 2002a).

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 2002b). 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 (130 km) and dives are deeper (frequently greater than 250 m). Summer foraging dives, however, are closer to shore (about 16 km) and shallower (100-250 m; (Loughlin 2002b; 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 of 70-140 m over 1-2 minutes, but much deeper dives in excess of 300 m are known (Loughlin et al. 2003; Merrick and Loughlin 1997; Rehberg et al. 2001). Young animals also tend to stay in shallower water less than 100 m deep and within 20 km from shore (Fadely et al. 2005).

Acoustics and hearing. Males and females apparently have different hearing sensitivities, with males hearing best at 1-16 kHz (best sensitivity at the low end of the range) and females hearing from 16-25 kHz (best hearing at the upper end of the range)(Kastelein 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 the 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.

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. The NMFS currently estimates the western DPS to have 42,366 individuals (Allen and Angliss 2010).

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 a 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).

The eastern stock seems to be performing better 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 fewer 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 recorded 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).

The NMFS considers this population stable or increasing, and multiplies pup counts by a factor of 4.5 (based on (Calkins and Pitcher 1982b) 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 ranges from 58,334-72,223 non-pup individuals (Allen and Angliss 2010). The 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).

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 suggests 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).

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.

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, presumably 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 (and their 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). PCB levels in the kidneys of some adult males are high enough that reproductive and immune function may have been compromised (Wang et al. 2011). 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. Steller sea lion contaminants are of additional concern 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 as well as major haulouts in Alaska and includes a 37 km buffer around these locations. 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.

Listed fishes

Puget Sound Chinook salmon

Description of the species. Chinook salmon are the largest of the Pacific salmon and historically ranged from the Ventura River in California to Point Hope, Alaska in North America, and in northeastern Asia from Hokkaido, Japan to the Anadyr River in Russia (Healey 1991a). In addition, Chinook salmon have been reported in the Canadian Beaufort Sea (McPhail and Lindsey 1970). We discuss the distribution, status, and critical habitats of the species[2] of endangered and threatened Chinook salmon separately, and summarize their common dependence on waters of the United States. However, because Chinook salmon in the wild are virtually indistinguishable between listed species, and are the same biological species we begin this section describing those characteristics common across the listed species.

Of the Pacific salmon species considered herein, Chinook salmon exhibit arguably one of the most diverse and complex life history strategies with multiple races within which there is substantial variation. One form, the "stream-type," resides in freshwater for a year or more following emergence and the "ocean-type" migrates to the ocean within their first year. The ocean-type typifies populations north of 56° N (Healey 1991a). Within each race, there is often variation in age at seaward migration, age of maturity, timing of spawning migrations, male precocity, and female fecundity.

Distribution. The Puget Sound Chinook salmon ESU includes all naturally spawned populations of Chinook salmon from rivers and streams flowing into Puget Sound including the Straits of Juan De Fuca from the Elwha River, eastward, with rivers and streams flowing into Hood Canal, South Sound, North Sound and the Strait of Georgia in Washington (Figure 2). Twenty-six artificial propagation programs are part of the ESU. These artificially propagated populations are no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU.

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[2] We use the word "species" as it has been defined in section 3 of the ESA, which include "species, subspecies, and any distinct population segment of any species of vertebrate fish or wildlife which interbreeds when mature (16 U.S.C 1533)." Pacific salmon that have been listed as endangered or threatened were listed as "evolutionarily significant units (ESU)" which NMFS uses to identify distinct population segments (DPS) of Pacific salmon. Any ESU or DPS is a "species" for the purposes of the ESA.

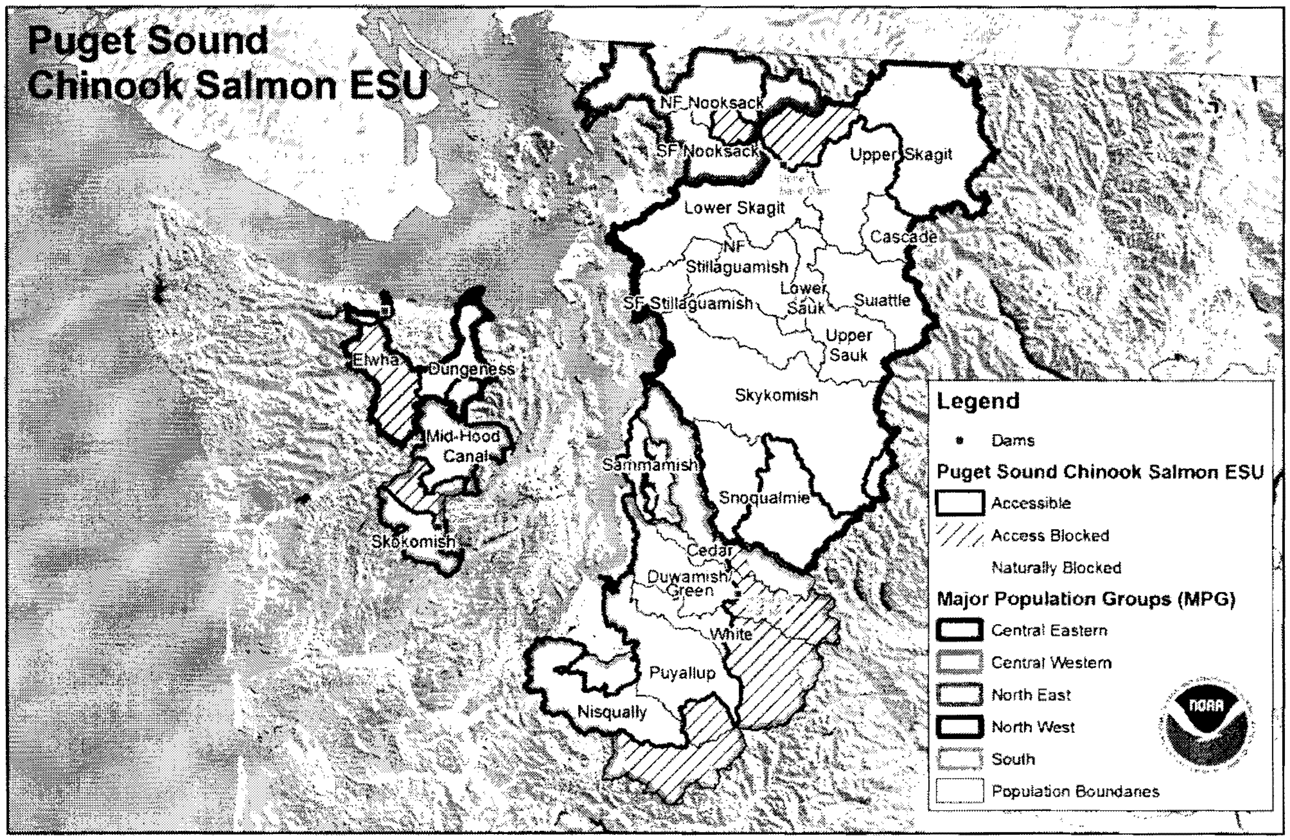

Figure 2. Population boundaries, dams, accessible areas, and extirpated reaches of Puget Sound Chinook distribution, incorporating critical habitat boundaries (dark black line). Taken from NMFS (NMFS 2011a).

The Puget Sound ESU is comprised of 31 historical populations, of which 22 or more are believed to be extant and nine are considered extinct. Table 7 identifies the current populations within the ESU for which there are data, and their recent abundance as well as long-term trends.

Chinook salmon in this area generally have an "ocean-type" life history. Puget Sound populations include both early-returning and late-returning spawners described by Healey (1991b). However, within these generalized behavioral forms, significant variation occurs in residence time in freshwater and estuarine environments. For example, Hayman et al. (1996) described three juvenile Chinook salmon life histories with varying residency times in the Skagit River system in northern Puget Sound. Chinook salmon utilize nearshore Puget Sound habitats year-round, although they can be far from their natal river systems (Brennan et al. 2004).

Table 7. Puget Sound Chinook salmon populations and selected measures of population viability.

Population	Historical abundance[a]	Mean number of spawners (natural-origin)[b]	Percent hatchery contribution (range)[c]	λ (+/- SE)[d]
Nooksack-North Fork	26,000	1,538 (125)	91 (88-95)	0.75 (0.07)
Nooksack-South Fork	13,000	338 (197)	40 (24-55)	0.94 (0.05)
Lower Skagit	22,000	2,527 (2,519)	0.2 (0-0.7)	1.05 (0.09)

Population	Historical abundance[a]	Mean number of spawners (natural-origin)[b]	Percent hatchery contribution (range)[c]	λ (+/- SE)[d]
Upper Skagit	35,000	9,489 (9,281)	2 (2-3)	1.05 (0.06)
Upper Cascade	1,700	274 (274)	0.3	1.06 (0.05)
Lower Sauk	7,800	601 (601)	0	1.01 (0.12)
Upper Sauk	4,200	324 (324)	0	0.96 (0.06)
Suiattle	830	365 (365)	0	0.99 (0.06)
Stillaguamish-North Fork	24,000	1,154 (671)	40 (13-52)	0.92 (0.04)
Stillaguamish-South Fork	20,000	270		0.99 (0.02)*
Skykomish	51,000	4,262 (2,392)	40 (11-66)	0.87 (0.03)
Snoqualmie	33,000	2,067(1,700)	16 (5-72)	1.00 (0.04)
North Lake Washington		331		1.07 (0.07)*
Cedar		327		0.99 (0.07)*
Green		8,884 (1,099)	83 (35-100)	0.67 (0.06)*
White		844		1.16 (0.06)*
Puyallup	33,000	1,653		0.95 (0.06)*
Nisqually	18,000	1,195		1.04 (0.07)*
Skokomish		1,392		1.04 (0.04)*
Dosewallips	4,700	48		1.17 (0.10)*
Duckabush		43		
Hamma Hamma		196		
Mid Hood Canal		311		
Dungeness	8,100	222		1.09 (0.11)*
Elwha		688		0.95 (0.11)*

[a] Estimated total historical abundance for this ESU was about 700,000 fish, but is not meant to reflect a summation of individual river historic estimates. Individual river estimates of historical abundance are based on an EDT analysis as reported in Good et al. 2005.

[b] 5-year geometric mean number of spawners (hatchery plus natural) for years 1998-2002. Geometric mean of natural origin spawners noted in parentheses. From Good et al. 2005.

[c] Percent hatchery-origin from 1997-2001. Estimates are from the TRT database and reported in Good et al. 2005.

[d] Short-term median population growth rate estimates assume that the reproductive success of naturally spawning hatchery fish is equivalent to that of natural origin fish. Except estimates noted * where an estimate of the fraction of hatchery fish is not available then λ represents hatchery fish + natural-origin spawners. Data years used for calculation 1990-2002 (Good et al. 2005).

Habitat. The time necessary for egg incubation until emergence of alevins in freshwater varies among basins and among years within a basin, and is closely correlated to water temperatures such that low temperatures can prolong incubation. Incubation generally takes a couple of months or more. Alevin (also called "yolk-sac" fry) remain buried until their yolk-sac is absorbed, at which time they become free swimming fry. Egg to fry survival can also vary widely across basins, years, and habitat conditions within a basin. In general, the survival of eggs and alevin, and the fitness of emerging fry are affected by sediment loading, intergravel water flow and dissolved oxygen levels, gravel composition, spawn timing, floods, redd and spawner density, and water temperatures.

Once emerged, fry behavior varies among populations and among individuals within races. Some juvenile Chinook salmon rear in freshwater for a few weeks to a few years, others move immediately downstream to coastal waters where they rear in estuaries for a few weeks to months, while others migrate directly to ocean waters. Stream-type Chinook salmon do not migrate to sea until the spring following emergence, and ocean-type Chinook salmon migrate to the ocean within their first year. Generally, most fry move at night probably to reduce detection by predators, although some fish will move downstream during daylight. Not all movement is volitional as stream flows often displace fry to downstream areas after emergence. Density-dependent factors such as space, prey, or stream flows may influence the outmigration behavior of individual juvenile Chinook salmon.

While in fresh water, juvenile Chinook salmon are often found in the lower reaches of a river near its estuary, where they inhabit river margins in areas of shallow water, near woody debris, or other areas of low water velocity. As juveniles grow in size, they tend to move away from the shoreline to deeper waters where the velocity is higher (Healey 1991a). The transformation from the freshwater fry/parr juvenile stage to smolt involves multiple physiological changes including increases in: body silvering, hypoosmotic regulatory capability, salinity tolerance and preference, growth rate, oxygen consumption, ammonia production, endocrine activity (e.g., activation of thyroid, interregnal and pituitary growth hormone cells), and gill Na^+, K^+-ATPase activity. At the same time, body condition declines (Wedemeyer et al. 1980). Several factors can affect the smoltification process, not only at the interface between freshwater and saltwater, but higher in the watershed as the process of transformation begins long before fish enter saltwaters, including exposure to heavy metals and elevated water temperatures (Wedemeyer et al. 1980).

Life at sea varies according to population, race, and age-class. Chinook salmon tend to remain at sea between 1 and 6 years, with most fish returning to freshwater after 2-4 years at sea. Fishery catches indicate that ocean- and stream-type fish exhibit divergent migratory pathways while in the ocean (Healey 1983; Healey 1991a). Ocean-type Chinook salmon tend to be found along the coastline, whereas stream-type Chinook salmon are found in the open ocean far from the coast (Healey 1983; Healey 1991a). Juvenile Chinook along the Pacific northwest, Canada, and Alaska tend to remain within roughly 55 km (most within 28 km) of the coast (NPFMC 1990; PFMC 2000). However, Chinook generally remain within 320 km of the coast (NPFMC 1990). Concentrations are known to occur around transient upwelling features (PFMC 2000). Individuals hatching in rivers south of Cape Blanco, Oregon tend to stay south of this point, while those outmigrating north of this point tend to move north into the Gulf of Alaska through coastal migratory corridors (PFMC 2000).

Generally, Chinook salmon outmigrants (termed smolts) are about 5-13 cm long when they enter saline (often brackish) waters. The process of smoltification is physiologically demanding, involving osmoregulation—the maintenance of osmotic pressure as the fish enters waters of increased salinity, which is necessary to maintain body fluid concentration and composition to maintain homeostasis. Smaller fish tend to remain closer to shore, while larger fish will enter marine waters. The distribution of fish in this stage does not appear to be correlated with salinities, as most fish can survive immediate transfer from freshwater to saltwater (~30 ppm salinity; (Healey 1991a). Once in the ocean, juveniles occupy waters 30-70 m deep and frequently associate with bottom topography (PFMC 2000). Although temperature ranges vary from 1° to 15° C, few individuals are found in waters below 5° C (MBC 1987; PFMC 2000). Chinook salmon originating from the same freshwater region have similar age-dependent marine distributions (even if associated with different runs), which are distinct from the distributions of Chinook from other freshwater regions (Weitkamp 2009). This distribution does not appear to be influenced by oceanographic variability (Weitkamp 2009). Older individuals appear to disperse more broadly than their younger counterparts (Weitkamp 2009). Adults do not appear to exhibit a preference in substrates and may be found down to 250 m (Beauchamp et al. 1983).

Reproduction. The general Chinook salmon life cycle spans fresh and marine waters, with one reproductive event per adult (Chinook salmon are semelparous and die after spawning). Spawning migrations generally occur in the spring and fall, although the precise timing of spawning migrations and spawning varies across populations and can vary within populations. Temperature and stream flow can significantly influence the timing of upstream migrations and spawning, and the selection of spawning habitat (Geist et al. 2009; Hatten and Tiffan 2009). For Klamath River Chinook, temperatures above 21.9° C (mean average body temperature), 20.6° C (mean minimum daily body temperature), or 23.1° C (mean maximum daily body temperature) completely inhibited upstream spawning migration in rivers; these values are close to the upper lethal limits for this and other salmonid species (Strange 2010). .A general latitudinal cline is apparent across the species range with spawning typically occurring earlier in the spring/summer at northern latitudes and later in southern latitudes (Healey 1991a).

On the spawning grounds, mate competition is intense with males competing to fertilize eggs and females competing for optimal nest site selection. Once fertilization occurs, female Chinook salmon bury the eggs in nests –termed "redds"- and guard the nests until their death, which generally occurs a couple days later to a couple weeks after spawning. A female generally deposits eggs in more than one depression within a redd, excavating stream rock as she moves upstream, increasing the size of her redd until all eggs are deposited.

Size and age at maturity is partially under genetic control, but can be influenced by environment and migration behavior (Roni and Quinn 1995). Generally, ocean-type salmon are at sea longer than their stream-type counterparts and tend to be larger in size at spawning. Body size can be important in determining reproductive success in terms of nest selection and mating competition (Foote 1990). Chinook salmon age at maturity ranges from 1 to 7 years with most returning to spawn between 3 and 4 years of age.

Feeding. Chinook salmon feed on a variety of prey organisms depending upon life stage. Adult oceanic Chinook salmon eat small fish, amphipods, and crab megalops (Brodeur et al. 2010)(Healey 1991). Fish, in particular herring, make up the largest portion of an adult Chinook salmon's diet. In estuaries, Chinook salmon smolts tend to feed on the chironomid larvae and pupae *Daphnia*, *Eogammarus*, *Corphium* and *Neomysis*, as well as juvenile herring, sticklebacks and other small fish. In freshwater, Chinook salmon juveniles feed on adult and larval insects including terrestrial and aquatic insects such as dipterans, beetles, stoneflies, chironomids, and plecopterans (Healey 1991a). During the first year of marine life, Puget Sound Chinook transition from nearshore foraging on insects and amphipods during June to offshore foraging on crab larvae and fish, such as herring during July through September (Duffy et al. 2010).

Status and trends. The NMFS listed Puget Sound Chinook salmon as threatened in 1999 (64 FR 14308); that status was reaffirmed on June 28, 2005 (70 FR 37160). This ESU has lost 15 spawning aggregations (nine from the early-run type) that were either independent historical populations or major components of the remaining 22 existing independent historical populations identified (Good et al. 2005b). The disproportionate loss of early-run life history diversity represents a significant loss of the evolutionary legacy of the ESU.

Data reported by Good et al. (2005) indicate that long term trends in abundance for this ESU are split with about half of the populations declining, and the other half increasing. In contrast, the short-term trend for four populations is declining. The overall long-term trend in abundance indicates that, on average, populations are just replacing themselves. Estimates of the short-term median population growth rate (λ)(1990-2002) indicate an even split between populations that are growing and those that are declining, although estimates would be lower for several populations if the fraction of naturally spawning hatchery fish were available for all populations within the ESU. For available data, when λ is calculated assuming that hatchery fish have the equivalent success of natural spawners then the largest estimated decline occurs in the Green River. Populations with the largest positive short and long-term trends include the White River and the North Fork Nooksack River (Good et al. 2005b). Lambda for the Skagit River, which produces the most Chinook salmon in this ESU, has increased slightly. Overall, the recent analysis by Good et al. (2005) illustrated that there has not be much change in this ESU since NMFS' first status review (Busby et al. 1996a). Individual populations have improved, while others have declined. However, the lack of information on the fraction of naturally spawning, hatchery-origin fish for 10 of the 22 populations within this ESU limits our understanding of the trends in naturally spawning fish for a large portion of the ESU. Natural-origin pre-harvest recruit escapements remained fairly constant from 1985-2009 (Ford et al. 2010). Returns (pre-harvest run size) from the natural spawners were highest in 1985, declined through 1994, remained low through 1999, increased in 2000 and again in 2001, and have declined through 2009, with 2009 having the lowest returns since 1997. Productivity in the five-year period from 2005-2009 has been the lowest in the ESU for any five-year period since 1985 and diversity of the populations has continued to decline; presently it is the lowest in the last 25 years (NMFS 2011a). Based on a Shannon Diversity Index at the ESU level, diversity is declining (due primarily to the increased abundance of returns to the Whidbey Basin region) for both distribution among populations and among regions (Ford et al. 2010).

The estimated total run size of Chinook salmon in Puget Sound in the early 1990s was 240,000 fish, representing a loss of nearly 450,000 fish from historic numbers. During a recent 5-year period, the geometric mean of natural spawners in populations of Puget Sound Chinook salmon ranged from 222 to just over 9,489 fish. Most populations had natural spawners numbering in the hundreds (median recent natural escapement is 766), and of the six populations with greater than 1,000 natural spawners, only two have a low fraction of hatchery fish. The populations with the greatest estimated component of hatchery fish tend to be in mid- to southern Puget Sound, Hood Canal, and the Strait of Juan de Fuca regions. Estimates of the historical equilibrium abundance, based on pre-European settlement habitat conditions, range from 1,700 to 51,000 potential spawners per population. The historical estimates of spawner capacity are several orders of magnitude higher than spawner abundances currently observed throughout the ESU (Good et al. 2005b).

Critical habitat. The NMFS designated critical habitat for Puget Sound Chinook salmon on September 2, 2005 (70 FR 52630). The specific geographic area includes portions of the Nooksack River, Skagit River, Sauk River, Stillaguamish River, Skykomish River, Snoqualmie River, Lake Washington, Green River, Puyallup River, White River, Nisqually River, Hamma Hamma River and other Hood Canal watersheds, the Dungeness/Elwha Watersheds, and nearshore marine areas of the Strait of Georgia, Puget Sound, Hood Canal, and the Strait of Juan de Fuca. This designation includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high water line is not defined the lateral extent is defined as the bank full elevation.

The designation for this ESU includes sites necessary to support one or more life stages. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. Specific primary constituent elements include freshwater spawning sites, freshwater rearing sites, freshwater migration corridors, nearshore marine habitat, and estuarine areas. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity. Of 49 subbasins reviewed in NMFS' assessment of critical habitat for the Puget Sound ESUs, nine subbasins were rated as having a medium conservation value, 12 were rated as low, and the remaining subbasins (40), where the bulk of Federal lands occur for this ESU, were rated as having a high conservation value to Puget Sound Chinook salmon. Factors contributing to the downward trends in this ESU are hydromorphological changes (such as diking, revetments, loss of secondary channels in floodplains, widespread blockages of streams, and changes in peak flows), degraded freshwater and marine habitat affected by agricultural activities and urbanization, and upper river tributaries widely affected by poor forest practices. Changes in habitat quantity, availability, diversity, flow, temperature, sediment load, and channel stability are common limiting factors in areas of critical habitat.

Landslides can occur naturally in steep, forested lands, but inappropriate land use practices likely have accelerated their frequency and the amount of sediment delivered to streams. Fine sediment from unpaved roads has also contributed to stream sedimentation. Unpaved roads are widespread on forested lands in the Puget Sound basin, and to a lesser extent, in rural residential areas. Historical logging removed most of the riparian trees near stream channels. Subsequent agricultural and urban conversion permanently altered riparian vegetation in the river valleys, leaving either no trees, or a thin band of trees. The riparian zones along many agricultural areas are now dominated by alder, invasive canary grass and blackberries, and provide substantially reduced stream shade and large wood recruitment (SSPS 2007).

Diking, agriculture, revetments, railroads and roads in lower stream reaches have caused significant loss of secondary channels in major valley floodplains in this region. Confined main channels create high-energy peak flows that remove smaller substrate particles and large wood. The loss of side-channels, oxbow lakes, and backwater habitats has resulted in a significant loss of juvenile salmonid rearing and refuge habitat. When the water level of Lake Washington was lowered 9 feet in the 1910s, thousands of acres of wetlands along the shoreline of Lake Washington, Lake Sammamish and the Sammamish River corridor were drained and converted to agricultural and urban uses. Wetlands play an important role in hydrologic processes, as they store water which ameliorates high and low flows. The interchange of surface and groundwater in complex stream and wetland systems helps to moderate stream temperatures. Forest wetlands are estimated to have diminished by one-third in Washington State ((FEMAT) 1993; Spence et al. 1996; SSPS 2007).

Loss of riparian habitat, elevated water temperatures, elevated levels of nutrients, increased nitrogen and phosphorus, and higher levels of turbidity, presumably from urban and highway runoff, wastewater treatment, failing septic systems, and agriculture or livestock impacts, have been documented in many Puget Sound tributaries (SSPS 2007).

Peak stream flows have increased over time due to paving (roads and parking areas), reduced percolation through surface soils on residential and agricultural lands, simplified and extended drainage networks, loss of wetlands, and rain-on-snow events in higher elevation clear cuts (SSPS 2007). In urbanized Puget Sound, there is a strong association between land use and land cover attributes and rates of coho spawner mortality likely due to runoff containing contaminants emitted from motor vehicles (Feist et al. 2011).

Juvenile mortality occurs in unscreened or inadequately screened diversions. Water diversion ditches resemble side channels in which juvenile salmonids normally find refuge. When diversion headgates are shut, access back to the main channel is cut off and the channel goes dry. Mortality can also occur with inadequately screened diversions from impingement on the screen, or mutilation in pumps where gaps or oversized screen openings allow juveniles to get into the system (WDFW 2009). Blockages by dams, water diversions, and shifts in flow regime due to hydroelectric development and flood control projects are major habitat problems in many Puget Sound tributary basins (SSPS 2007).

The nearshore marine habitat has been extensively altered and armored by industrial and residential development near the mouths of many of Puget Sound's tributaries. A railroad runs along large portions of the eastern shoreline of Puget Sound, eliminating natural cover along the shore and natural recruitment of beach sand (SSPS 2007).

Dams constructed for hydropower generation, irrigation, or flood control have substantially affected Puget Sound Chinook salmon populations in a number of river systems. The construction and operation of dams have blocked access to spawning and rearing habitat (*e.g.*, Elwha River dams block anadromous fish access to 70 miles of potential habitat) changed flow patterns, resulted in elevated temperatures and stranding of juvenile migrants, and degraded downstream spawning and rearing habitat by reducing recruitment of spawning gravel and large wood to downstream areas (SSPS 2007)). These actions tend to promote downstream channel incision and simplification (Kondolf 1997), limiting fish habitat. Water withdrawals reduce available fish habitat and alter sediment transport. Hydropower projects often change flow rates, stranding and killing fish, and reducing aquatic invertebrate (food source) productivity (Chappell 1980).

Natural threats. Chinook salmon are exposed to high rates of natural predation during freshwater rearing and migration stages, as well as during ocean migration. In general, Chinook salmon are prey for pelagic fishes, birds, and marine mammals, including harbor seals, sea lions, and killer whales. There have been recent concerns that the increasing size of tern, seal, and sea lion populations in the Pacific northwest may have reduced the survival of some salmon populations; 10% of salmonid smolts are eaten by Caspian terns and double-crested cormorants annually in the Columbia River estuary (NMFS 2011b). Invasive fishes also threaten the survival and recovery of Pacific salmonids by competing directly for resources, altering food webs and trophic structures, and altering evolutionary trajectories (NMFS 2011b).

Anthropogenic threats. Chinook salmon have declined under the combined effects of fishery over-harvest; competition from fish raised in hatcheries and native and non-native exotic species; dams that block their migrations and alter river hydrology; gravel mining that impedes their migration and alters the dynamics (hydrogeomorphology) of the rivers and streams that support juveniles; water diversions that deplete water levels in rivers and streams; destruction or degradation of riparian habitat that increase water temperatures in rivers and streams sufficient to reduce the survival of juvenile Chinook salmon; and land use practices (logging, agriculture, urbanization) that destroy wetland and riparian ecosystems while introducing sediment, nutrients, biocides, metals, and other pollutants into surface and ground water and degrade water quality in the freshwater, estuarine, and coastal ecosystems throughout the Pacific northwest (Buhle et al. 2009). Of several habitat factors for spring-run Columbia River Chinook salmon, reductions in fine sediments may be particularly important to changes in population numbers (Honea et al. 2009).

Population declines have resulted from several human-mediated causes, but the greatest negative influence has likely been the establishment of waterway obstructions such as dams, power plants, and sluiceways for hydropower, agriculture, flood control, and water storage. These structures have blocked salmon migration to spawning habitat or resulted in direct mortality and have eliminated entire salmon runs as a result. While some of these barriers remain, others have been reengineered, renovated, or removed to allow for surviving runs to access former habitat, but success has been limited. These types of barriers alter the natural hydrograph of basins, both upstream and downstream of the structure, and significantly reduce the availability and quality of spawning and rearing habitat (Hatten and Tiffan 2009). Many streams and rivers, particularly in urban or suburban areas, suffer from streamside development, which contributes sediment, chemical pollutants from pesticide applications and automobile or industrial activities, altered stream flows, loss of streamside vegetation and allochthonous materials to name a few. These factors can directly cause mortality, reduce reproductive success, or affect the health and fitness of all salmon life stages.

Artificial propagation of hatchery fish has had profound consequences on the viability of some natural salmon populations, but there are potential benefits to the artificial production of salmon as well. Adverse effects of artificial propagation include: a decline in the natural population from the taking of wild brood stock for artificial propagation, the genetic erosion of populations (introgression, hybridization), an increase incidence of disease to and increased rates of competition with and predation on naturally spawned salmon populations. Potential benefits to artificial propagation include the bolstering of the numbers of naturally spawning fish in the short-term, the conservation of genetic resources, and guarding against the catastrophic loss of naturally spawned populations at critically low abundance levels.

Fishing for salmon has also negatively impacted salmon populations. Fishing reduces the number of individuals within a population and can lead to uneven exploitation of certain populations and size classes (Mundy 1997; Reisenbichler 1997). Targeted fishing of larger individuals results in excluding the most fecund individuals from spawning (Reisenbichler 1997). Genetic changes that promote smaller body sizes have occurred in heavily exploited populations in response to size-selective harvest pressures (Mundy 1997; Reisenbichler 1997; Swain et al. 2007). Age at maturity can also be accelerated by fishing pressure (Reisenbichler 1997). Pacific salmon species are exposed to a number of contaminants throughout their range and life history cycle.

Exposure to pollution is also of significant concern for all life stages, but is likely particularly significant for freshwater life stages. Organic pollutants, especially PCBs, DDT and its congeners, pesticides, and endocrine disruptor,s are particularly concerning. These chemicals can inhibit smell, disrupt reproductive behavior and physiology, impair immune function, and lead to mortality through impairment of water balance when traveling between fresh- and saltwater systems (Varanasi et al. 1993a; Varanasi et al. 1993b). Diffuse and extensive population centers contribute increase contaminant volumes and variety from such sources as wastewater treatment plants and sprawling development. Urban runoff from impervious surfaces and roadways often contains oil, copper, pesticides, PAHs, and other chemical pollutants and flow into surface waters. Point and nonpoint pollution sources entering rivers and their tributaries affect water quality in available spawning and rearing habitat for salmon. Juvenile salmonids that inhabit urban watersheds often carry high contaminant burdens, which is partly attributable to the biological transfer of contaminants through the food web (Brown et al. 1985; Stein et al. 1992; Varanasi et al. 1993a).

Hood Canal summer-run chum salmon

Description of the species. Chum salmon are more widely distributed than other salmon and may have at one time made up nearly 50% of the Pacific salmon biomass in the Pacific Ocean (Salo 1991). Historically, chum salmon were distributed throughout the coastal regions of western Canada and the United States, as far south as Monterey Bay, California, to the Arctic coast and east to the Mackenzie River, in the Beaufort Sea. They also ranged in Asia from Korea to the Arctic coast of Russia and west to the Lena River. Presently, major spawning populations on the west coast of the United States are found only as far south as Tillamook Bay on the northern Oregon coast. In this section of the Opinion, we discuss the distribution, status, and critical habitats of the two listed species of threatened chum salmon separately; however, because chum salmon in the wild are virtually indistinguishable between listed ESUs, and are the same biological species sharing the same generalized life history, we begin this section describing those characteristics common across ESUs.

Chum salmon exhibit obligatory anadromy (there are no recorded landlocked or naturalized freshwater populations), and like Chinook salmon, chum salmon are semelparous (die after one spawning event). Their general life cycle spans fresh and marine waters, although chum salmon are more marine oriented than other Pacific salmon in that they spend very little time rearing in freshwater. Chum salmon spend 2-5 years in feeding areas in the northeast Pacific Ocean, which is a greater proportion of their life history than other Pacific salmonids. Chum salmon distribute throughout the North Pacific Ocean and Bering Sea, although North American chum salmon (as opposed to chum salmon originating in Asia), rarely occur west of 175° E longitude. North American chum salmon migrate north along the coast in a narrow coastal band that broadens in southeastern Alaska, although some data suggest that Puget Sound chum, including Hood Canal summer run chum, may not make extended migrations into northern British Columbian and Alaskan waters, but instead may travel directly offshore into the north Pacific Ocean.

Distribution. The Hood Canal summer-run chum salmon ESU includes all naturally spawned populations of summer-run chum salmon in Hood Canal and its tributaries as well as populations in Olympic Peninsula rivers between Hood Canal and Dungeness Bay, Washington (64 FR 14508)(Figure 3) from mid-September to mid-October (WDF (Washington Department of Fisheries) 1993), but may enter natal rivers in late August. Eight artificial propagation programs are considered to be part of the ESU. The NMFS determined that these artificially propagated populations are no more divergent relative to the local natural population(s) than what would be expected between closely related natural populations within the species. Table 8 identifies populations within the Hood Canal summer-run chum salmon ESU, their abundances, and hatchery input.

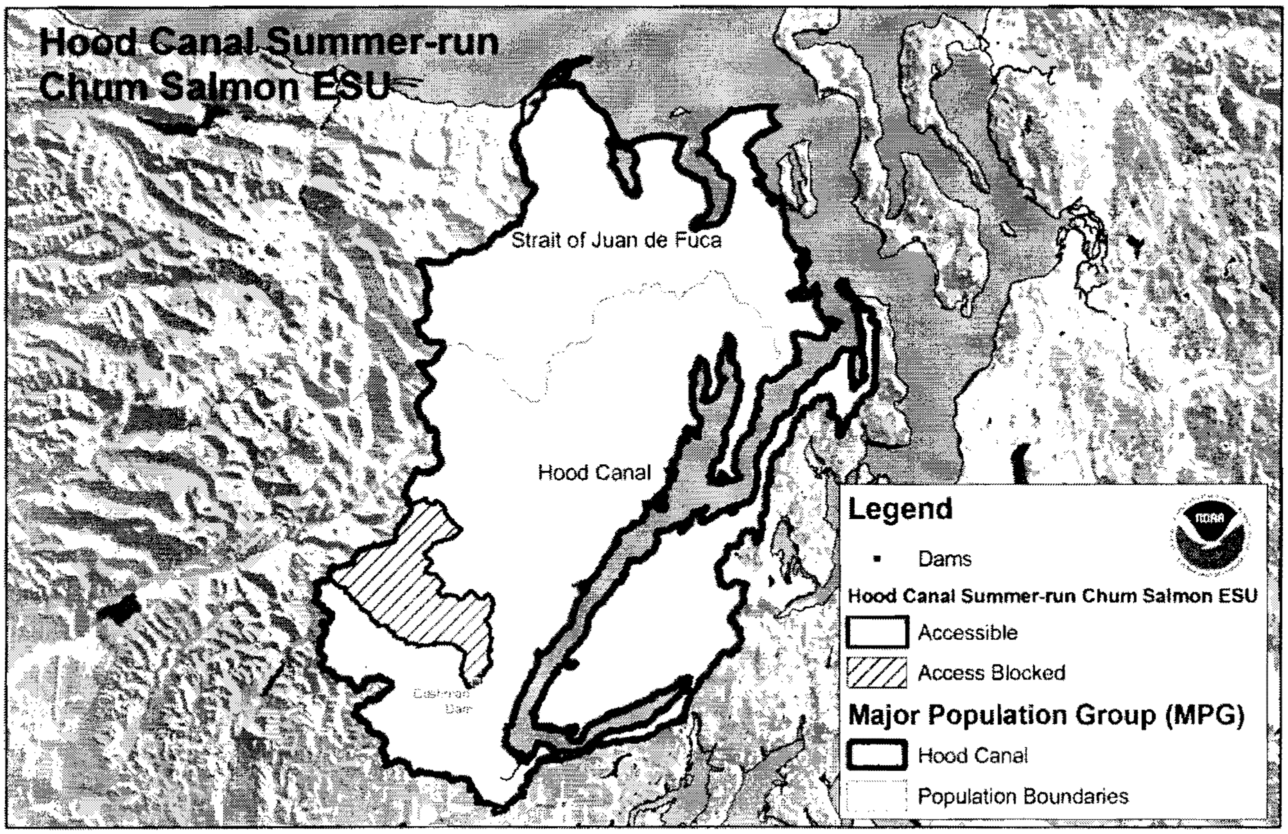

Figure 3. Population boundaries, dams, accessible areas, and extirpated reaches of Hood Canal summer-run chum distribution, incorporating critical habitat boundaries (dark black line). Taken from NMFS (NMFS 2011a).

Table 8. Hood Canal summer-run chum populations and selected measures of population viability.

Populations[a]	**1999-2002 mean escapement (range)**	**Percent hatchery contributions (1995-2001)**	**λ (+/- SE)**
Jimmycomelately Creek	10 (1-192)		0.85 (0.16)
Salmon/Snow creeks	1,521 (463-5,921)	0-69	1.23 (0.10)
Big/Little Quilcene rivers	4,512 (3,065-6,067)	5-51	1.39(0.22)
Lilliwaup Creek	13 (1-775)		1.19 (0.44)
Hamma Hamma River	558 (173-2,260)		1.3 (0.19)
Duckabush River	382 (92-942)		1.1 (0.17)
Dosewallips River	919 (351-1,627)		1.17 (0.24)
Union River			1.15 (0.10)
Chimacum Creek*	198 (0-903)[c]	100	
Big Beef Creek*	17 (0-826)[c]	100	
Dewatto Creek*	9 (2-32)[d]		

[a] All data is reported in Good et al. 2005. *Denotes extinct populations that have recently had some natural recolonization or have been seeded with hatchery fish.

On average Hood Canal chum salmon reside in estuaries for 23 days; daily tidal migrations have not been observed, but prey availability does influence movement patterns (Bax 1983). Upon leaving their natal estuaries, individuals generally migrate through Hood Canal and into the main body of Puget Sound.

Habitat. Chum salmon are found in freshwater to euryhaline water at depths ranging from the surface to 250 m, although juveniles are primarily epipelagic and are found from the surface down to 95 m and within 36 km of shore (Emmett et al. 1991; Salo 1991). Chum salmon are found at a wide range of temperatures from 3° to 22° C but prefer temperatures from 8.3° to 15.6° C (Pauley et al. 1988). Juveniles migrate within the Gulf of Alaska coastal belt during their first summer at sea (Salo 1991). Maturing individuals are also distributed widely in the Gulf of Alaska during spring and summer (Salo 1991).

Chum salmon usually spawn in the lower reaches of rivers, with redds usually dug in the mainstem or in side channels of rivers from just above tidal influence to roughly 80 km upstream. Juveniles outmigrate to seawater almost immediately after emerging from the gravel that covers their redds (Salo 1991). This ocean-type migratory behavior contrasts with the stream-type behavior of some other species in the genus *Oncorhynchus* (e.g., coastal cutthroat trout, steelhead, Coho salmon, and most types of Chinook and sockeye salmon), which usually migrate to sea at a larger size, after months or years of freshwater rearing. This means that survival and growth in juvenile chum salmon depend less on freshwater conditions (unlike stream-type salmonids which depend heavily on freshwater habitats) than on favorable estuarine conditions. Another behavioral difference between chum salmon and species that rear extensively in freshwater is that chum salmon form schools, presumably to reduce predation (Pitcher 1986), especially if their movements are synchronized to swamp predators (Miller and Brannon 1982).

Reproduction. Spawning migrations generally occur in the summer and fall; the precise spawn timing and migration varies across populations. Stream flows and water temperatures can influence stream entry. Sexual differences in the timing of returns to spawning grounds are apparent, with males generally arriving early and females later in the run. Once on the spawning grounds mate competition is intense with males competing to fertilize eggs and females competing for optimal nest site selection. Size and age at maturity is partially under genetic control, but can be influenced by environment and migration behavior. Generally, spawning runs consist of fish between 2 and 5 years of age, and like Chinook salmon, chum females will build large redds that consist of four or five egg pockets laid in succession. Chum salmon fecundity is highly variable, and is correlated with body size and region (latitudinal trends are evident with northern population having lower absolute and relative fecundities)(Salo 1991)).

Size and age at maturity is partially under genetic control, but can be influenced by environment and migration behavior. Generally, spawning runs consist of fish between 2 and 5 years of age, and like Chinook salmon, chum females will build large redds that consist of 4 or 5 egg pockets (Salo 1991). Chum salmon fecundity is highly variable, and is correlated with body size and region (latitudinal trends are evident with northern population having lower absolute and relative fecundities; (Salo 1991).

The time necessary for egg incubation until emergence of alevins in freshwater varies among basins and among years within a basin, and is closely correlated to water temperatures such that low temperatures prolong incubation. Egg and alevin survival, and the fitness of emerging fry are affected by sediment loading, intergravel water flow and dissolved oxygen levels, gravel composition, spawning time and density, and water temperatures.

Once they emerge from their gravel nests, chum salmon fry outmigrate to seawater almost immediately (Salo 1991).

Feeding. Generally, chum fry emigrate to estuaries between March and May where they forage on epibenthic and neritic food resources. As food resources decline and the fish grow, they move further out to forage on pelagic and nektonic organisms (Salo 1991; Simenstad and Salo 1982). The timing of juvenile entry into seawater is commonly correlated with nearshore warming and associated plankton blooms (Groot and Margolis 1991). General migratory studies indicate that chum salmon in their first year of life will typically maintain a coastal migratory pattern although the pattern is variable as they mature at sea. At sea, chum salmon feed on pteropods, euphausiids, amphipods, fish, and squid larvae (Salo 1991). Chum salmon spend two to five years in feeding areas in the northeast Pacific Ocean, which is a greater proportion of their life history than other Pacific salmonids.

Status and trends. The NMFS listed Hood Canal summer-run chum salmon as threatened on March 25, 1999 (64 FR 14508), and reaffirmed this status on June 28, 2005 (70 FR 37160). Historically, Hood Canal summer-run chum salmon comprised an estimated 16 populations; only eight extant populations remain (Good et al. 2005b). Most of the extirpated populations historically occurred on the eastern side of Hood Canal, which is cause for concern over the current spatial structure of this ESU. The widespread loss of estuary and lower floodplain habitat is a continuing threat to ESU spatial structure and connectivity.

Although some population returns showed modest improvements in 2000, with upward trends continuing in 2001 and 2002, the recent 5-year mean abundance is variable, ranging from one fish to nearly 4,500 fish in the Big/Little Quilcene rivers. Hood Canal summer-run chum are the focus of an extensive rebuilding program developed and implemented since 1992 by state and tribal comanagers. Two populations (the combined Quilcene and Union River populations) are above the conservation thresholds established by the rebuilding plan. However, most populations remain depressed. Estimates of the fraction of naturally spawning hatchery fish exceed 60% for some populations, indicating that reintroduction programs are supplementing the numbers of total fish spawning naturally in streams. Long-term trends in productivity are above replacement for only the Quilcene and Union River populations. Buoyed by recent increases, seven populations are exhibiting short-term productivity trends above replacement. Although the 1994-2004 productivity trend for most populations was increasing, that trend generally reversed from 2005-2009 (NMFS 2011a).

Of the eight programs releasing individuals considered to be part of the ESU, six of the programs are supplementation programs implemented to preserve and increase the abundance of native populations in their natal watersheds. The NMFS' assessment of the effects of artificial propagation on ESU extinction risk concluded that these hatchery programs collectively do not substantially reduce the extinction risk of the ESU. The hatchery programs are reducing risks to ESU abundance by increasing total ESU abundance as well as the number of naturally spawning individuals.

Critical habitat. The NMFS designated critical habitat for Hood Canal summer-run chum salmon on September 2, 2005 (70 FR 52630). The specific geographic area includes the Skokomish River, Hood Canal subbasin (Hamma Hamma and Dosewallips rivers and others), the Puget Sound subbasin, Dungeness/Elwha subbasin, and nearshore marine areas of Hood

Canal and the Strait of Juan de Fuca from the line of extreme high tide to a depth of 30 meters. This includes a narrow nearshore zone from the extreme high tide to mean lower low tide within several Navy security/restricted zones. This also includes about 8 miles of habitat that was unoccupied at the time of the designation (Finch, Anderson and Chimacum creeks; 69 FR 74572; 70 FR 52630), but has recently been re-seeded. Chimacum Creek, however, has been naturally recolonized since at least 2007 (T. Johnson, pers. comm., Jan. 2010). The designation for Hood Canal summer-run chum, like others made at this time, includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high-water line is not defined the lateral extent is defined as the bank full elevation.

The specific primary constituent elements identified for Hood Canal summer-run chum salmon are areas for spawning, freshwater rearing and migration, estuarine areas free of obstruction, nearshore marine areas free of obstructions, and offshore marine areas with good water quality. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity. Of 17 subbasins reviewed in NMFS' assessment of critical habitat for the ESU, 14 subbasins were rated as having a high conservation value, while only three were rated as having a medium value to the conservation. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. Limiting factors identified for this species include degraded floodplain and mainstem river channel structure, degraded estuarine conditions and loss of estuarine habitat, riparian area degradation and loss of in-river wood in the mainstems, excessive sediment in spawning gravels, and reduced stream flow in migration areas.

Degradation of the near-shore environment has occurred in the southeastern areas of Hood Canal in recent years, resulting in late summer marine oxygen depletion and significant fish kills. Circulation of marine waters is naturally limited, and partially driven by freshwater runoff, which is often low in the late summer. However, human development has increased nutrient loads from failing septic systems along the shoreline, and from use of nitrate and phosphate fertilizers on lawns and farms. Shoreline residential development is widespread and dense in many places. The combination of highways and dense residential development has degraded certain physical and chemical characteristics of the near-shore environment (Brewer et al. 2007; SSPS 2007).

Natural threats. Chum salmon are exposed to high rates of natural predation at each life stage, particularly during migration. Mortality at or prior to emergence is significant because eggs develop in the interstitial spaces in the stream gravel; storm surges that redeposit gravel and wash out eggs or introduce silt to the interstitial spaces can reduce egg survival. Other factors that reduce egg survival and larvae development include low dissolved oxygen, poor percolation, and extreme cold or warm temperatures. In freshwater, fry fall prey to older salmon and other trout, as well as birds, sculpin, and various mammals; 10% of salmonid smolts are eaten by Caspian terns and double-crested cormorants annually in the Columbia River estuary (NMFS 2011b). Invasive fishes also threaten the survival and recovery of Pacific salmonids by competing directly for resources, altering food webs and trophic structures, and altering evolutionary trajectories (NMFS 2011b).

Anthropogenic threats. Chum salmon have declined under the combined effects of overharvests in fisheries; competition from fish raised in hatcheries and native and non-native exotic species; dams that block their migrations and alter river hydrology; gravel mining that impedes their migration and alters the dynamics (hydrogeomorphology) of the rivers and streams that support juveniles; water diversions that deplete water levels in rivers and streams; destruction or degradation of riparian habitat that increase water temperatures in rivers and streams sufficient to reduce the survival of juvenile chum salmon; and land use practices (logging, agriculture, urbanization) that destroy wetland and riparian ecosystems while introducing sediment, nutrients, biocides, metals, and other pollutants into surface and ground water and degrade water quality in the freshwater, estuarine, and coastal ecosystems throughout the Pacific northwest.

Puget Sound steelhead

Description of the species. Steelhead, the common name of the anadromous form of *O. mykiss*, are native to Pacific Coast streams extending from Alaska south to northwestern Mexico (Good et al. 2005a; Good et al. 2005b; Moyle 1976; NMFS 1997; Stolz and Schnell 1991). The life history of this species varies considerably throughout its range. Generally, steelhead occur in two races: the stream-maturing type, summer steelhead, enters freshwater in a sexually immature condition and requires several months in freshwater to mature and spawn; and the ocean-maturing type, winter steelhead, enters freshwater with well-developed gonads and spawns shortly after river entry. Variations in migration timing exist between populations, and some river basins have both summer and winter steelhead, while others only have one race.

There is a high degree of overlap in spawning timing between populations regardless of run type (Busby et al. 1996b). Difficult field conditions at that time of year and the remoteness of spawning grounds contribute to the relative lack of specific information on steelhead spawning. Unlike Pacific salmon, steelhead are iteroparous, or capable of spawning more than once before death (Busby et al. 1996b; Nickelson et al. 1992). Second-time spawners often make up about 70-85% of repeat spawners, with third time spawners make up 10-25% of repeats (Stolz and Schnell 1991). Iteroparity is more common among southern steelhead populations than northern populations (Busby et al. 1996b).

Distribution. Puget Sound steelhead occupy river basins of the Strait of Juan de Fuca, Puget Sound, and Hood Canal, Washington (Figure 4). Included are river basins as far west as the Elwha River and as far north as the Nooksack River. Puget Sound's fjord-like structure may affect steelhead migration patterns; for example, some populations of Coho and Chinook salmon, at least historically, remained within Puget Sound and did not migrate to the Pacific Ocean itself. Even when Puget Sound steelhead migrate to the high seas, they may spend considerable time as juveniles or adults in the protected marine environment of Puget Sound, a feature not readily accessible to steelhead from other areas of the Pacific northwest. This species is primarily composed of winter steelhead but includes several stocks of summer steelhead, usually in subbasins of large river systems and above seasonal hydrologic barriers. Life history attributes of Puget Sound steelhead (migration and spawn timing, smolt age, ocean age, and total age at first spawning) appear similar to those of other west coast steelhead.

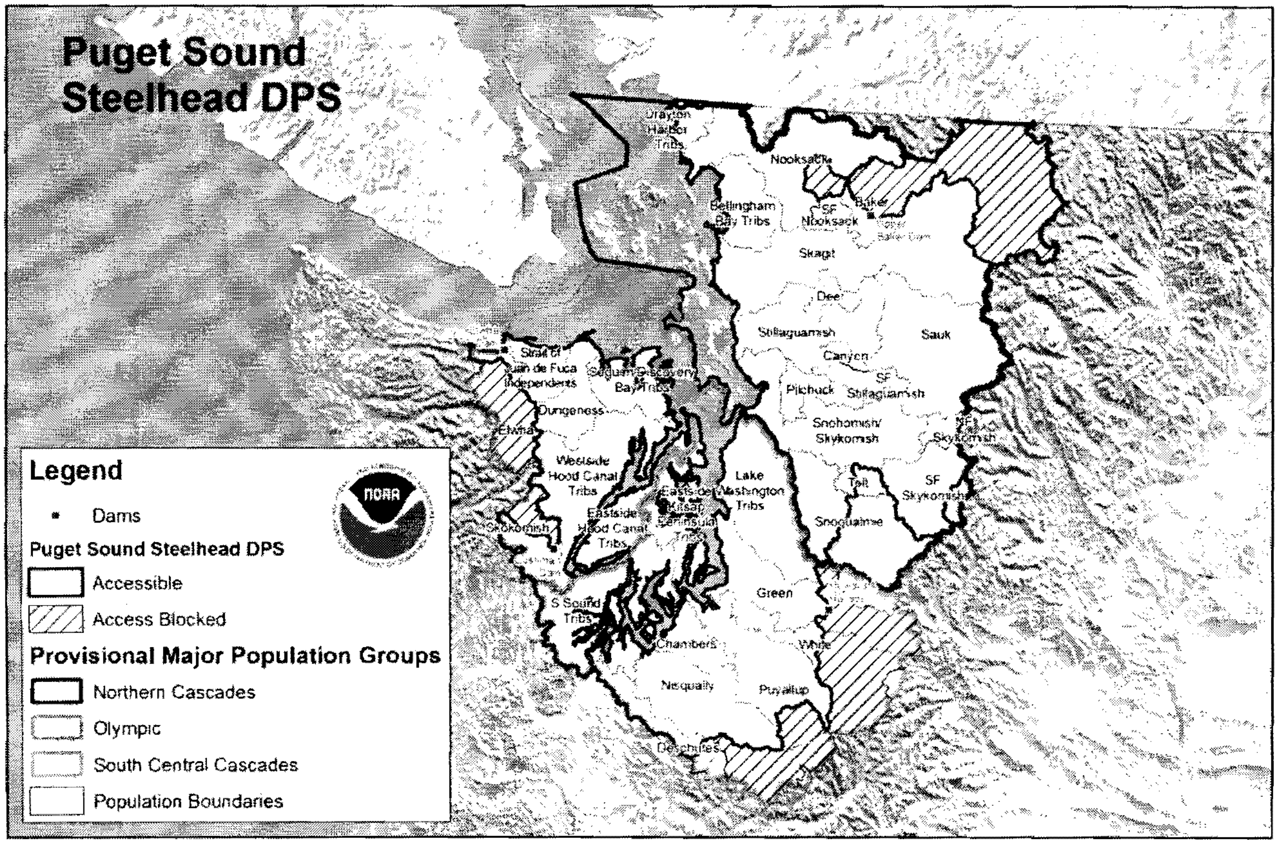

Figure 4. Population boundaries, dams, accessible areas, and extirpated reaches of Puget Sound steelhead distribution, incorporating critical habitat boundaries (dark black line). Taken from NMFS (NMFS 2011a).

Habitat. Steelhead occur in marine waters from the surface down to 200 m in waters with temperatures up to 24° C, although 10° C is optimum (Pauley et al. 1986).

Summer steelhead enter freshwater between May and October in the Pacific northwest (Busby et al. 1996b; Nickelson et al. 1992). They require cool, deep holding pools during summer and fall, prior to spawning (Nickelson et al. 1992). Summer steelhead migrate inland toward spawning areas, overwinter in the larger rivers, resume migration in early spring to natal streams, and then spawn in January and February (Barnhart 1986; Meehan and Bjornn 1991; Nickelson et al. 1992). Winter steelhead enter freshwater between November and April in the Pacific northwest (Busby et al. 1996b; Nickelson et al. 1992), migrate to spawning areas, and then spawn, generally in April and May (Barnhart 1986). Some adults, however, do not enter some coastal streams until spring, just before spawning (Meehan and Bjornn 1991).

As with other salmonids, the larger the fish, the more eggs produced. Egg and hatching success are related to the conditions within the redd and time to hatching is temperature dependent. Fertilization to hatching is generally less than a month, after which newly hatched fish will remain in the redd for another 2-3 weeks. In late spring and following yolk sac absorption, alevins emerge from the gravel and begin actively feeding. After emerging from the gravel, fry usually inhabit shallow water along banks of perennial streams. Fry occupy stream margins (Nickelson et al. 1992). Summer rearing takes place primarily in the faster parts of pools, although young-of-the-year are abundant in glides and riffles. Winter rearing occurs more uniformly at lower densities across a wide range of fast and slow habitat types. Some older juveniles move downstream to rear in larger tributaries and mainstem rivers (Nickelson et al. 1992).

Juvenile steelhead migrate little during their first summer and occupy a range of habitats featuring moderate to high water velocity and variable depths (Bisson et al. 1988). Steelhead hold territories close to the substratum where flows are lower and sometimes counter to the main stream; from these, they can make forays up into surface currents to take drifting food (Kalleberg 1958). Juveniles rear in freshwater from 1 to 4 years, then smolt and migrate to the ocean in March and April (Barnhart 1986). Winter steelhead juveniles generally smolt after 2 years in freshwater (Busby et al. 1996b). Juvenile steelhead tend to migrate directly offshore during their first summer from whatever point they enter the ocean rather than migrating along the coastal belt as salmon do. Steelhead typically reside in marine waters for 2 or 3 years prior to returning to their natal stream to spawn as 4- or 5-year olds; fish in the northern portion of the range may spend more time rearing in marine waters (Stolz and Schnell 1991).

Feeding. Juveniles feed primarily on insects (chironomids, baetid mayflies, and hydropsychid caddisflies (Merz 1994). Adults feed on aquatic and terrestrial insects, mollusks, crustaceans, fish eggs, minnows, and other small fishes (including greenling and other trout; (Chapman and Bjornn 1969; Stolz and Schnell 1991)). Survival at smoltification is higher for larger fish than smaller ones; this is particularly true for individuals that grew larger earlier in life (Beakes et al. 2010).

Mortality. Steelhead mortality is high early in life and decreases with age. For example, Puget Sound steelhead leaving freshwater and estuarine habitats experience 55-86% survival to the point of reaching Hood Canal and 0-49% from Hood Canal to the Strait of Juan de Fuca, with survival increasing greatly upon entering the Pacific Ocean (Moore et al. 2010).

Status and trends. The DPS was listed as a threatened species on May 11, 2007 (72 FR 26722). Run size was calculated in the early 1980s at about 100,000 winter-run fish and 20,000 summer-run fish. It is not clear what portion were hatchery fish, but a combined estimate with coastal steelhead suggested that roughly 70% of steelhead in ocean runs were of hatchery origin. The percentage in escapement to spawning grounds would be substantially lower due to differential harvest and hatchery rack returns. By the 1990s, total run size for four major stocks exceeded 45,000; roughly half of which was natural escapement.

Nehlsen et al. (1991) identified nine Puget Sound steelhead stocks at some degree of risk or concern, while the WDFW et al. (1993) estimated that 31 of 53 stocks were of native origin and predominantly natural production. Their assessment of the status of these 31 stocks was 11 healthy, three depressed, one critical, and 16 of unknown status. Their assessment of the status of the remaining (not native/natural) stocks was three healthy, 11 depressed, and eight of unknown status.

Of the 21 populations in the Puget Sound ESU reviewed by Busby et al. (1996b), 17 had declining and four increasing trends, with a range from 18% annual decline (Lake Washington winter-run steelhead) to 7% annual increase (Skykomish River winter-run steelhead). These trends were for the late-run naturally produced component of winter-run steelhead populations; no adult trend data were available for summer-run steelhead. Most of these trends were based on relatively short data series. The Skagit and Snohomish River winter-run populations have been approximately three to five times larger than the other populations in the DPS, with average annual spawning of approximately 5,000 and 3,000 total adult spawners, respectively. These two basins exhibited modest overall upward trends at the time of the Busby et al. (1996b) report. Busby et al. (1996b) estimated five-year average natural escapements for streams with adequate data range from less than 100 to 7,200, with corresponding total run sizes of 550 to 19,800. NMFS (2011a) and Ford et al. (2010) reported that all but a few small populations were currently declining at a 3-10% rate annually, with a high risk of extinction in the next 100 years (Table 9).

Table 9. Puget Sound steelhead populations and risk of extinction (Ford et al. 2010).

Geographic region (MPGs)	Population (Watershed)	Extinction risk (probability of decline to 10% of its current estimated abundance)
Northern Cascades	Samish River (winter)	High—about 80% within 25 years
	Skagit River (winter)	High—about 80% within 75 years.
	Snohomish River (winter)	Moderately High—about 50% within 100 years
	Stillaguamish River (winter)	High—about 90% within 60 years
	Tolt River summer	High—nearly 80% within 100 years
	Nooksack River (winter)	Unable to calculate
South Puget Sound	Lake Washington (winter)	High—~ 90% within 40 years
	Green River (winter)r	High—about 90% within 80 years
	Nisqually River (winter)	High—about 80% within 40 years
	Puyallup River (winter)	High—about 90% within 25-30 years
	White River (winter)	High—about 90% within 50 years
	South Sound Tributaries (winter)	Unable to calculate
Olympic	Elwha River (winter)	Fairly High—~ 90% within 40 years
	Dungeness River (winter)	High—within 100 years (population too low to calculate %)
	Port Angeles (winter)	High—nearly 80% within 100 years
	West Hood Canal (winter)	Low—near zero within 100 years
	East Hood Canal (winter)	Low—about 30% within 100 years
	Skokomish River (winter)	High—about 80% within 80 years

Threats. Steelhead are exposed to high rates of natural predation each life stage. The highest mortality occurs between the egg stage and smolt outmigration, and is highest in the first few months following emergence from the redd (Stolz and Schnell 1991). In freshwater, fry fall prey to older steelhead and other trout, as well as birds, sculpin, and various mammals; 10% of salmonid smolts are eaten by Caspian terns and double-crested cormorants annually in the Columbia River estuary (NMFS 2011b). In the ocean, marine mammals and other fish prey on steelhead, but the extent of such predation is not well known. Invasive fishes also threaten the survival and recovery of Pacific salmonids by competing directly for resources, altering food webs and trophic structures, and altering evolutionary trajectories (NMFS 2011b).

Steelhead have declined under the combined effects of overharvests in fisheries; competition from fish raised in hatcheries and native and non-native exotic species; dams that block their migrations and alter river hydrology; gravel mining that impedes their migration and alters the dynamics (hydrogeomorphology) of the rivers and streams that support juveniles; water diversions that deplete water levels in rivers and streams; destruction or degradation of riparian habitat that increase water temperatures in rivers and streams sufficient to reduce the survival of juvenile steelhead; and land use practices (logging, agriculture, urbanization) that destroy wetland and riparian ecosystems while introducing sediment, nutrients, biocides, metals, and other pollutants into surface and ground water and degrade water quality in the fresh water, estuarine, and coastal ecosystems throughout the species range.

Critical habitat. All steelhead critical habitat was published on September 2, 2005 (70 FR 52488). Critical habitat has been designated for all DPSs except Puget Sound steelhead. All steelhead critical habitat includes the same PCEs for the same conservation reasoning:

- Freshwater spawning sites with water and substrate quantity to support spawning, incubation, and larval development.
- Freshwater rearing sites with water quantity and floodplain connectivity to form and maintain physical habitat conditions and support juvenile growth, foraging, behavioral development (e.g., predator avoidance, competition), and mobility. Specific features include forage supporting juvenile development as well as natural cover such as shade, submerged and overhanging large wood, log jams and beaver dams, aquatic vegetation, large rocks and boulders, side channels, and undercut banks.
- Freshwater migration corridors free of obstruction with water quantity and natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, side channels, and undercut banks to support mobility and survival. Without these features, juveniles cannot avoid high flows and predators, successfully compete, begin the behavioral and physiological changes needed for marine life, or out-migrate.
- Estuarine areas free of obstruction with water quality and salinity conditions supporting juvenile and adult physiological transitions between fresh- and saltwater. Estuaries must also include natural cover (submerged and overhanging large wood, aquatic vegetation, large rocks and boulders), side channels as well as prey for both juveniles and adults. These features are essential to conservation because without them juveniles cannot reach the ocean in a timely manner and use the variety of habitats that allow them to avoid predators, compete successfully, and complete the behavioral and physiological changes needed for life in the ocean.
- Nearshore marine areas free of obstruction with water quality and quantity conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation; and natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels. No areas are specifically designated as critical habitat, but areas under this category are an important component to Oregon Coast Coho life history.

- Offshore marine areas with water quality conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation. These features are essential for conservation because without them juveniles cannot forage and grow to adulthood.

Species-specific threats and limitations to recovery. Numerous impacts hinder the survival and recovery of Puget Sound steelhead (NMFS 2011a). These include widespread declines in adult abundance (total run size), despite significant reductions in harvest in recent years and use of two hatchery steelhead stocks (Chambers Creek and Skamania) inconsistent with wild stock diversity throughout the DPS. Further impairment results from declining diversity in the DPS, including the uncertain but weak status of summer-run fish in the DPS, a reduction in spatial structure for steelhead in the DPS, and reduced habitat quality through changes in river hydrology, temperature profile, downstream gravel recruitment, and reduced movement of large woody debris. Further habitat-based threats include increased flood frequency and peak flows during storms, reduced groundwater-driven summer flows in the lower reaches of many rivers and their tributaries in Puget Sound where urban development has occurred, has resulted in gravel scour, bank erosion, and sediment deposition as well as dikes, hardening of banks with riprap, and channelization, which have reduced river braiding and sinuosity, have increased the likelihood of gravel scour and dislocation of rearing juveniles.

Bocaccio

Description of the species. The bocaccio is a rockfish species that genetic analyses suggest is composed of two distinct populations (Matala et al. 2004; Wishard et al. 1980). A southern population exists along the Pacific coasts of Mexican and California and is separated from a northern population by a region of apparent scarcity from northern California to southern Oregon (MacCall and He 2002b). It has been proposed that oceanographic features, such as current patterns restricting larval movement, are responsible for population discreteness (Matala et al. 2004; NMFS 2008d). The northern population is the entity that is proposed for listing. However, the presence of a third population has also been suggested (Queen Charlotte Island, Vancouver Island to Point Conception, California, and south of Point Conception)(Matala et al. 2004). For stock management purposes, the NMFS and Pacific Fisheries Management Council recognize these populations as separate stocks.

Distribution. Bocaccio occur from the central Baja peninsula of Mexico north along the continental shelf and slope as far as Stepovac Bay, Alaska (Love et al. 2002).

Habitat and movement. Preferred bocaccio habitat is largely dependent upon the life stage of an individual. Larvae and young juveniles tend to be found in deeper offshore regions (1-148 km offshore), but associated with the surface and occasionally with floating kelp mats (Emery et al. 2006; Hartmann 1987; Love et al. 2002). As individuals mature into older juveniles and adults, they transition into shallow waters and settle to the bottom, preferring algae-covered rocky, eelgrass, or sand habitats and aggregating into schools (Eschmeyer et al. 1983; Love et al. 1991). After a few weeks, fish move into slightly deeper waters of 18-30 m and occupy rocky reefs (Carr 1983; Eschmeyer et al. 1983; Feder et al. 1974; Johnson 2006; Love and Yoklavich 2008). As adults, bocaccio may be found in depths of 12-478 m, but tend to remain in shallow waters on the continental shelf (20-250 m), still associating mostly with reefs or other hard substrate, but may move over mud flats (Feder et al. 1974; Kramer and O'Connell 1995; Love et al. 2005; Love et al. 2006; Love et al. 2002; Love and York 2005). Artificial habitats, such as platform structures, also appear to be suitable habitat for bocaccio (Love and York 2006).

Adults may occupy territories of 200-400 hectares, but can venture outside of this territory (Hartmann 1987). Adults tend to occupy deeper waters in the southern population compared to the northern population (Love et al. 2002). Adults are not as benthic as juveniles and may occur as much as 30 m above the bottom and move 100 m vertically during the course of a day as they move between different areas (Love et al. 2002; Starr et al. 2002). Prior to severe population reductions, bocaccio appeared to frequent the Tacoma Narrows in Washington State (DeLacy et al. 1964; Haw and Buckley 1971; Miller and Borton 1980).

Reproduction. Bocaccio are live-bearers with internal fertilization. Once females become mature (at 54-61 cm total length), they produce 20,000-2.3 million eggs annually, with the number increasing as females age and grow larger (Echeverria 1987; Hart 1973; Love et al. 2002). However, either sex has been known to attain sexual maturity as small as 35 cm or 3 years of age and, in recent years as populations have declined, average age at sexual maturity may have declined as well (Echeverria 1987; Hart 1973; Love et al. 2002; MacCall 2002b). Mating occurs between August and November, with larvae born between January and April (Love et al. 2002; Lyubimova 1965; MacCall and He 2002b; Moser 1967; Westrheim 1975; Wyllie Echeverria 1987).

Growth. Upon birth, bocaccio larvae measure 4-5 mm in length. These larvae move into pelagic waters as juveniles when they are 1.5-3 cm and remain in oceanic waters from 3.5-5.5 months after birth (usually until early June), where they grow at ~0.5-1 mm per day (Love et al. 2002; MacCall 2003; MacCall and He 2002b; Matarese et al. 1989; Moser 1967; Woodbury and Ralston 1991). However, growth can vary from year-to-year (Woodbury and Ralston 1991). Once individuals are 3-4 cm in length, they return to nearshore waters, where they settle into bottom habitats. Females tend to grow faster than males, but fish may take 5 years to reach sexual maturity (MacCall 2003). Individuals continue to grow until they reach maximum sizes of 91 cm, or 9.6 kg, at an estimated maximum age of 50 years (Andrews et al. 2005a; Eschmeyer et al. 1983; Halstead et al. 1990; Love et al. 2002; Piner et al. 2006; Ralston and Ianelli 1998). However, individuals tend to grow larger in more northerly regions (Dark et al. 1983).

Foraging. Prey of bocaccio vary with fish age, with bocaccio larvae starting with larval krill, diatoms, and dinoflagellates (Love et al. 2002). Pelagic juveniles consume fish larvae, copepods, and krill, while older, nearshore juveniles and adults prey upon rockfishes, hake, sablefish, anchovies, lanternfish, and squid (Love et al. 2002; Reilly et al. 1992).

Acoustics and hearing. Data regarding bocaccio hearing are not available. However, field measurements have recorded bocaccio calls, which are more prevalent at night, as <900 Hz repetitive pulses of ~0.1 s duration (Sirovic et al. 2009a).

Status and trends. Bocaccio were proposed for listing on April 23, 2009 (74 FR 18516). Bocaccio as a species has undergone severe decline in the past several decades, with the species currently estimated to be 3.6% of its abundance in 1970 (MacCall and He 2002b). Prior to World War II, commercial landings of rockfish species generally remained under 20,000 lbs, but sky-rocketed during the war to 375,000 lbs annually and fluctuated between 50,000 and 220,000 lbs until 1970, when landings increased linearly with fishing effort to a peak of 900,000 lbs by 1980 (Palsson et al. 2008). Levels fluctuated after this between 48,000 and 300,000 lbs for the next decade and clearly crashed in the 1990's, with landings below 30,000 lbs annually. At the cessation of commercial fishing in 2003, 2,600 lbs of rockfish were harvested. Similar trends are seen in recreational landings from Puget Sound (WDF 1975-1986).

Among rockfish of the Puget Sound, bocaccio appear to have undergone a particular decline (MacCall and He 2002b). This has likely because of the removal of the largest, most fecund individuals of the population due to overfishing and the frequent failure of recruitment classes, possibly because of unfavorable climactic/oceanographic conditions (MacCall and He 2002b).

Bocaccio resistance to depletion and recovery is also hindered by demographic features (Love et al. 1998a). Bocaccio are long-lived fishes, taking several years to reach sexual maturity and becoming more fecund with age (Dorn 2002). As harvesting targeted the largest individuals available, bocaccio have become less capable of recovering population numbers (Love et al. 1998b). At present, in the complete absence of directed or bycatch fishing pressure, it is estimated that bocaccio populations would have to have frequent good recruitment to restrain their present decline (Tolimieri and Levin 2005). In addition, bocaccio reproduction appears to be characterized by frequent recruitment failures, punctuated by occasional high success years (Love et al. 1998b; MacCall and He 2002b). Over the past 30 years, 1977, 1984, and 1988 are the only years in which recruitment appears to have been significant successes (it should be noted that 1999 and 2002 also appear to have been strong, but survivorship into maturity is still pending). Recruitment success appears to be linked to oceanographic/climactic patterns and may be related to cyclic warm/cool ocean periods, with cool periods having greater success (Love et al. 1998b; MacCall 1996; Moser et al. 2000b; Sakuma and Ralston 1995). Harvey et al. (2006) suggested that bocaccio may have recently diverted resources from reproduction, potentially resulting in additional impairment to recovery. Overall, bocaccio have the highest variability of recruitment of any rockfish studied to date, with recruitment exhibiting a random walk and high temporal variability (MacCall and He 2002b; Tolimieri and Levin 2005).

Although population estimates are not available for the northern population, the southern population has been estimated to number 1.6 million fish of 1 year of age or older in 2002 (MacCall 2002a). Of these, 1.0 million were estimated to occur south of Pt. Conception, where recruitment has been stronger. However, individuals north of Pt. Conception tend to be larger and, hence, more fecund. In 2002, the southern population was estimated to produce 720 billion eggs annually (243 billion south of Pt. Conception). North of Pt. Conception, bocaccio are most abundant in the Monterey Bay area, where prime habitat seems to be over the continental slope and, secondarily, over the shelf (Dark et al. 1983).

The rate of decline for rockfish in Puget Sound has been estimated at ~3% annually for the period 1965-2007. Various rebuilding estimates for bocaccio populations have predicted recovery, but require long periods (98-170 years) and assume no mortality from fishing (intentional harvests are closed, but bycatch still occurs)(MacCall 2008; MacCall and He 2002a; NMFS 2008d).

Natural threats. Interspecies competition, predators, and climactic regimes are the primary natural factors that depress bocaccio numbers. Copper and quillback rockfish may compete with bocaccio in Puget Sound for available resources (NMFS 2008d). King salmon, lingcod, terns and other seabirds, harbor seals, and Steller sea lions are know predators of bocaccio and other rockfish species (Beaudreau and Essington 2007; Lance and Jeffries 2007; Love et al. 2002). Bocaccio and other rockfish appear to be negatively influenced by El Niño conditions, possibly reducing available prey supply (Harvey 2005; Moser et al. 2000a).

Anthropogenic threats. Although overfishing is the primary reason for bocaccio being proposed as a listed species, bycatch and habitat loss are also human-related factors that have likely led to bocaccio decline. Although a frequent species captured in fisheries during the late 1970's, bocaccio were not recorded from any recreational surveys from 1996-2007 (Palsson et al. 2008; WDF 1975-1986). Apart from commercial fishing, recreational fishing (even catch-and-release) appears to incur significant mortality on bocaccio and other rockfishes (Schroeder and Love 2002). The species is considered overfished by the Pacific Fisheries Management Council and is not presently harvested intentionally. However, bycatch is still considered to be a high impact stressor to rockfish populations of Washington State waters (Palsson et al. 2008).

Habitat loss is also a factor in bocaccio decline, with rocky habitats (reportedly, there are only 217 km^2 in Puget Sound) being threatened by construction of bridges, sewer lines, cable and pipeline deployment, and dredge spoil (Palsson et al. 2008). Loss of kelp, which is valuable to juvenile fish recruitment, as well as anoxic conditions, exacerbate habitat loss (NMFS 2008d).

Critical habitat. Critical habitat has not been proposed or designated for the bocaccio.

Yelloweye rockfish

Description of the species. Yelloweye rockfish are likely composed of at least two populations and possibly more. Yamanaka et al. (2006) found that those individuals found within the Georgia Basin and Queen Charlotte Strait were genetically distinct from other samples from Oregon to Alaska. The Georgia Basin/Queen Charlotte Sound population is the one which has been proposed for listing in U.S. waters.

Distribution. Yelloweye rockfish occur from Baja California to the Aleutian Islands, but are most common from central California to Alaska (Love et al. 2002).

Habitat. As with other rockfishes, yelloweye habitat varies based upon life stage. Larvae maintain a pelagic existence but as juveniles, move into shallow high relief rocky or sponge garden habitats (Eschmeyer et al. 1983; Love et al. 1991; Richards et al. 1985). Juveniles may also associate with floating debris or pilings (Lamb and Edgell 1986). As adults, yelloweye rockfish move in to deeper habitats. Individuals have been found in waters as deep as 549 m, but are generally found in waters of less than 180 m (Eschmeyer et al. 1983; Love et al. 2002). However, adults continue to associate with rocky, high relief habitats, particularly with caves and crevices, pinnacles, and boulder fields (Carlson and Straty 1981; Love et al. 1991; O'Connell and Carlisle 1993; Richards 1986; Yoklavich et al. 2000). Yelloweyes generally occur as individuals, with loose, residential aggregations infrequently found (Coombs 1979; DeMott 1983; Love et al. 2002). In the Puget Sound region, sport catch records from the 1970's indicate that Sucia Island and other islands of the San Juans as well as Bellingham Bay had the highest concentrations of catches (Delacy et al. 1972; Miller and Borton 1980).

Reproduction. Yelloweye rockfish are live bearers with internal fertilization. Copulation occurs between September and April, with fertilization taking place later as latitude increases (DeLacy et al. 1964; Hitz 1962; Lea et al. 1999; O'Connell 1987; Westrheim 1975; Wyllie Echeverria 1987). Puget Sound yelloweyes mate between winter and summer, giving birth from spring to late summer (Washington et al. 1978). Gestation lasts roughly 30 days (Eldridge et al. 2002). Although yelloweye rockfish were once believed to reproduce annually, evidence exists that indicate the potential for multiple births per year (MacGregor 1970; Washington et al. 1978). Females produce more eggs as they grow older and larger, with each individual producing roughly 300 eggs per year per gram of body weight (1.2-2.7 million eggs per year)(Hart 1973; MacGregor 1970). In addition, older females of several rockfish species may be capable of provisioning their offspring better than their younger counterparts, meaning that they may be more a more influential component in a given year's recruitment success (Sogard et al. 2008).

Growth and development. Larvae are born at 4-5 mm in length and maintain a pelagic existence for the first 2 months of life, before moving to nearshore habitats and settling into rocky reef habitat at about 25 mm in length (DeLacy et al. 1964; Love et al. 2002; Matarese et al. 1989; Moser 1996a). Yelloweye growth is thought to vary by latitudinal gradient, with individuals in more northerly regions growing faster and larger. Year class strength appears to be most strongly linked to survival of the larval stage (Laidig et al. 2007). In general, sexual maturity appears to be reached by 50% of individuals by 15-20 years of age and 40-50 cm in length (Yamanaka and Kronlund 1997). As with other rockfish, yelloweyes can be long-lived (reported oldest age is 118 years)(Munk 2001). Maximum size has been reported as 910 cm, but assymptotic size in Alaskan waters for both males and females was estimated to be 690 cm and 659-676 mm along British Columbia (Clemens and Wilby 1961; Love et al. 2005; Rosenthal et al. 1982; Westrheim and Harling 1975; Yamanaka et al. 2006).

Movement. Individuals shift to deeper habitats as they age. Juveniles tend to begin life in shallow rocky reefs and graduate to deeper rocky habitats as adults. Once adult habitat is established, individuals tend to remain at a particular site (Coombs 1979; DeMott 1983; Love 1978).

Foraging. As with other rockfish species, yelloweye rockfish prey upon different species and size classes throughout their development. Larval and juvenile rockfish prey upon phyto- and zooplankton (Lee and Sampson 2009). Adult yelloweyes eat other rockfish (including members of their own species), sand lance, gadids, flatfishes, shrimp, crabs, and gastropods (Love et al. 2002; Yamanaka et al. 2006).

Status and trends. Yelloweye rockfish were proposed for listing on April 23, 2009 (73 FR 18516). Yelloweye rockfish abundance has been variable in the Puget Sound region over the past 60 years, ranging from less than 1% to greater than 3% of samples, although Wallace (2001) documented large historical population in the Strait of Georgia. The latest samples have been historic lows in abundance. Perhaps more importantly, age classes appear to have been truncated to younger, smaller fish, severely hampering the ability of the species to recover from its primary cause of decline: overfishing (Berkeley et al. 2004).

Prior to World War II, commercial landings of rockfish species generally remained under 20,000 lbs, but sky-rocketed during the war to 375,000 lbs annually and fluctuated between 50,000 and 220,000 lbs until 1970, when landings increased linearly with fishing effort to a peak of 900,000 lbs by 1980 (Palsson et al. 2008). Levels fluctuated after this between 48,000 and 300,000 lbs for the next decade and clearly crashed in the 1990's, with landings below 30,000 lbs annually. At the cessation of commercial fishing in 2003, 2,600 lbs of rockfish were harvested. Over the period of 1965-2007, it is estimated that rockfish species has declined by 3% per year.

The most recent estimate of yelloweye rockfish abundance in the Puget Sound region was 3,000 individuals, with low abundance through spawning areas (Palsson et al. 2008).

Natural threats. Interspecies competition, predators, and climactic regimes are the primary natural factors that depress yelloweye rockfish numbers. Copper and quillback rockfish may compete with yelloweye rockfish in Puget Sound for available resources (NMFS 2008d). Lingcod, killer whales, and Steller sea lions are likely predators of yelloweye and other rockfish species (Beaudreau and Essington 2007; Lance and Jeffries 2007; Love et al. 2002). Yelloweye and other rockfish appear to be negatively influenced by El Niño conditions, possibly reducing available prey supply (Black 2009; Harvey 2005; Moser et al. 2000a). Oceanographic conditions (such as sea level anomalies and nearshore temperature conditions) appear to strongly influence the strength of each year's recruitment (Laidig et al. 2007). Rates of natural mortality have been reported to range from 2-4.6% annually (Wallace 2007; Yamanaka and Kronlund 1997).

Anthropogenic threats. Overfishing is considered the primary cause of yelloweye rockfish decline throughout their range, including in Washington State and British Columbian waters (NMFS 2008d; Wallace 2007). Although commercial harvesting of the species has ended, bycatch is still considered to be a high impact stressor to rockfish populations of Washington State waters (Palsson et al. 2008). It has been estimated that yelloweye rockfish have fallen 30% in abundance within 1/3 of a generation in the past few decades, an astonishing rate of decline.

Habitat loss is also a factor in yelloweye decline, with rocky habitats (reportedly, there are only 217 km^2 in Puget Sound) being threatened by construction of bridges, sewer lines, cable and pipeline deployment, and dredge spoil (Palsson et al. 2008). Anoxic conditions and chemical contamination are also considered threats to yelloweye rockfish recovery (NMFS 2008d).

Canary rockfish

Description of the species. It is unclear how many populations compose canary rockfish as a species. Genetic analysis have found that individuals south of Cape Blanco in southern Oregon lack an allele that individuals north of this point have (Wishard et al. 1980). This has been used to support the proposal of a northern DPS. In addition, canary rockfish are managed as two stocks in Canadian waters (COSEWIC in press). However, clear evidence of genetically or morphologically distinct populations is still lacking.

Distribution. Canary rockfish are found from the northern Baja peninsula north to the western Gulf of Alaska, and with the greatest abundance along British Columbia to central California (Cailliet et al. 2000; Hart 1973; Love et al. 2002; Miller and Lea 1972).

Habitat. Canary rockfish occupy a variety of habitats based upon their life stage. Larvae and younger juveniles tend to occupy shallow waters at the beginning of their lives, but generally remain in the upper 100 m of the water column (Love et al. 2002). Juveniles initially settle into tide pools and rocky reefs (Cailliet et al. 2000; Love et al. 1991; Love et al. 2002; Miller and Geibel 1973). Juveniles have also been observed in diurnal movements, occurring near sand-rock interfaces in groups by day and moving over sandy areas at night (Love et al. 2002). After as much as 3 years, juveniles move into deeper rocky reefs, forming loose schools, rarely on but generally near the bottom (Boehlert 1980; Cailliet et al. 2000; Johnson et al. 2003; Lamb and Edgell 1986; Methot and Stewart 2005; Phillips 1960; Rosenthal et al. 1998; Starr 1998; Tissot et al. 2007). Adults may be found in waters of up to 400 m, but tend to be most common in the 80-200 m range, or even shallower (Methot and Stewart 2005; Moser 1996b; Tissot et al. 2007). Mid shelf locations seem to have the highest concentrations of canary rockfish off Washington and Oregon (Weinberg 1994). Adults tend to occur in shallow areas in higher latitudes than their southern counterparts, although adults do appear to move into progressively deeper waters as they age (Methot and Stewart 2005; Vetter and Lynn 1997). It is believed that, within Puget Sound, canary rockfish were most common in the 1960's and 1970's in Tacoma Narrows, Hood Canal, San Juan Islands, Bellingham, and Appletree Cove (Delacy et al. 1972; Miller and Borton 1980). A latitudinal gradient may be present by age class, with older and larger individuals preferably occupying more northerly habitat (Dark et al. 1983).

Movement. Individual canary rockfish can range widely (up to 700 km over several years), although patterns of residency have been observed (Casillas et al. 1998; DeMott 1983; Gascon and Miller 1981; Lea et al. 1999; Love et al. 2002). In addition, seasonal movements have been found, with individuals moving from 160-210 m depths in late winter to 100-170 m in late summer (COSEWIC in press).

Reproduction. Canary rockfish develop their young internally before giving birth to live young as larvae. During each annual spawning event, a female can produce 260,000 to 1.9 million eggs, depending upon her size and age (Guillemot et al. 1985; NMFS 2008d). Unlike some other rockfish, there does not appear to be a latitudinal or geographic gradient associated with number of eggs produced (Gunderson et al. 1980; Love et al. 2002). Birth takes place in Oregonian and Washingtonian waters between September through March, with a peak in December and January. The peak in British Columbian waters is slightly later (February)(Barss 1989; Hart 1973; Westrheim and Harling 1975; Wyllie Echeverria 1987).

Growth and development. When born, larvae are 3.6-4.0 mm in length and take from 1-4 months to develop into juveniles (Krigsman 2000; Love et al. 2002; Moser 1996a; Richardson and Laroche 1979; Stahl-Johnson 1985; Waldron 1968). As with other rockfish, females seem grow more quickly than do males, with females reaching sexual maturity at 7-9 years of age (35-45 cm in length) versus males at 7-12 years (~41 cm in length) off Oregon (Boehlert and Kappenman 1980; Lenarz and Echeverria 1991; STAT 1999; Westrheim and Harling 1975). Mean length at sexual maturity off Vancouver Island is 41 cm for females and 48 cm for males (Westrheim and Harling 1975). Canary rockfish are known to frequently reach 60-75 years of age and have been found to be as old as 84 years (Andrews et al. 2007; Cailliet et al. 2001; Cailliet et al. 2000). Maximum reported sizes are 76 cm and 4.5 kg (Boehlert 1980; IGFA 1991; Love et al. 2002; Methot and Stewart 2005; Williams et al. 1999).

Foraging. Canary rockfish prey upon different species as they age. Larvae are planktivores, consuming invertebrate eggs, copepods, and nauplii (Love et al. 2002; Moser and Boehlert 1991). Juveniles feed upon zooplankton, including crustaceans, juvenile polychaetes barnacle cyprids, and euphasiid eggs and larvae (Gaines and Roughgarden 1987; Love et al. 1991). However, adults move into a carnivorous lifestyle as well as eating euphasiids and other crustaceans. Adults consume other fishes such as shortbelly rockfish, mytophids and stomiatiods (Cailliet et al. 2000; Love et al. 2002). However, oceanographic and climactic shifts can alter foraging such that canary rockfish feed on other available species (Lee and Sampson 2009).

Status and trends. Canary rockfish were proposed for listing on April 23, 2009 (74 FR 18516). Canary rockfish were once considered common in Puget Sound, but has declined at a faster rate than any other rockfish species in the region (Holmberg et al. 1967; NMFS 2008d). Prior to World War II, commercial landings of rockfish species generally remained under 20,000 lbs, but sky-rocketed during the war to 375,000 lbs annually and fluctuated between 50,000 and 220,000 lbs until 1970, when landings increased linearly with fishing effort to a peak of 900,000 lbs by 1980 (Palsson et al. 2008). Levels fluctuated after this between 48,000 and 300,000 lbs for the next decade and clearly crashed in the 1990's, with landings below 30,000 lbs annually. At the cessation of commercial fishing in 2003, 2,600 lbs of rockfish were harvested. Canary rockfish have been noted for being much less frequently caught in the Puget Sound and Georgia Basin region since 1965 (NMFS 2008d). The rate of decline for rockfish in Puget Sound has been estimated at ~3% annually for the period 1965-2007.

Declines have been noted in both numbers as well as frequencies. This likely due to the targeted removal of larger, older, and more fecund individuals by commercial fisheries, reducing the ability of canary rockfish to rebound from excessive mortality (NMFS 2008d). For example, recreational fishing data have not reported any individuals caught greater than 55 cm since 2000, whereas a variety of large size classes had formerly been caught. There are concerns that even now some populations have been lost entirely, primarily due to over harvesting, but also due to low dissolved oxygen levels in some areas of Puget Sound (NMFS 2008d).

Natural threats. Interspecies competition, predators, and climactic regimes are the primary natural factors that depress canary rockfish numbers. Copper and quillback rockfish may compete with canary rockfish in Puget Sound for available resources (NMFS 2008d). Predators of canary rockfish include other rockfishes, lingcod (for which rockfish is a particularly important dietary component), cabezon, seabirds, salmon, sharks, dolphins, seals, Steller sea lions, and perhaps river otters (Ainley et al. 1981; Antonelis Jr. and Fiscus 1980; Beaudreau and Essington 2007; Lance and Jeffries 2007; Love et al. 1991; Merkel 1957; Miller and Geibel 1973; Morejohn et al. 1978; Roberts 1979; Rosenthal et al. 1982; Stevens and Miller 1983). Canary and other rockfishes appear to be negatively influenced by El Niño conditions, possibly reducing available prey supply (Harvey 2005; Moser et al. 2000a).

Anthropogenic threats. Overharvesting the primary cause of canary rockfish declines, but habitat loss is also important. Canary rockfish are considered overfished by the Pacific Fisheries Management Council and are not presently harvested intentionally. However, bycatch is still considered to be a high impact stressor to rockfish populations of Washington State waters (Palsson et al. 2008). Habitat loss is also a factor in canary rockfish decline, with rocky habitats (reportedly, there are only 217 km^2 in Puget Sound) being threatened by construction of bridges, sewer lines, cable and pipeline deployment, and dredge spoil (Palsson et al. 2008). Low oxygen levels as well as pollutant, chemical, and nutrient loading are also considered significant threats to canary rockfish recovery (NMFS 2008d).

Critical habitat. Critical habitat has not been designated or proposed for canary rockfish.

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; Wearmouth and Sims 2008). However, satellite telemetry indicate average distance to shore is generally within 25 km (Baird et al. 2011). Individuals can move between islands within a matter of days (Wearmouth and Sims 2008). 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 timespan (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; Wearmouth and Sims 2008). 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 (Wearmouth and Sims 2008). Some prey, such as mahimahi, 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; Wearmouth and Sims 2008). 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; Shallenberger 1981a; Wearmouth and Sims 2008). 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 (Wearmouth and Sims 2008) 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 (Wearmouth and Sims 2008). Data from 1993-1998 support a population estimate of 121 individuals, which is likely negatively biased (Mobley Jr. et al. 2000; Wearmouth and Sims 2008). 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 (Wearmouth and Sims 2008).

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 (Wearmouth and Sims 2008). 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 (Wearmouth and Sims 2008). 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 2009b; 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 2009d). 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).

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 or proposed marine mammals in the action area.

Climate change

Climate change represents a significant threat to listed species and has contributed significantly to the extinctions of several species of diverse taxa around the planet (Monzón et al. 2011). 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 0.74° 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 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, Galapagos sea lion adult and pup, and overall harbor porpoise survival (Forcada et al. 2005; Macleod et al. 2007; Palacios et al. 2011). 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), southern resident killer whales, and Hawaiian insular false killer whales. 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 (Monzón et al. 2011; 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 2008f).

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 affect 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 2004). The acoustic noise that commercial traffic contributes to the marine environment is a concern for listed species because it may impair communication between individuals or cause animals to avoid certain areas (Brumm 2010; 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. 1995b). 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. 1995b). 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). Sonars and small vessels contribute significantly to mid-frequency ranges (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.

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.

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 2008f). 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 2008f).

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 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. 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 whales (Figure 5). 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.

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).

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 2005; Carretta et al. 2004a).

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). 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).

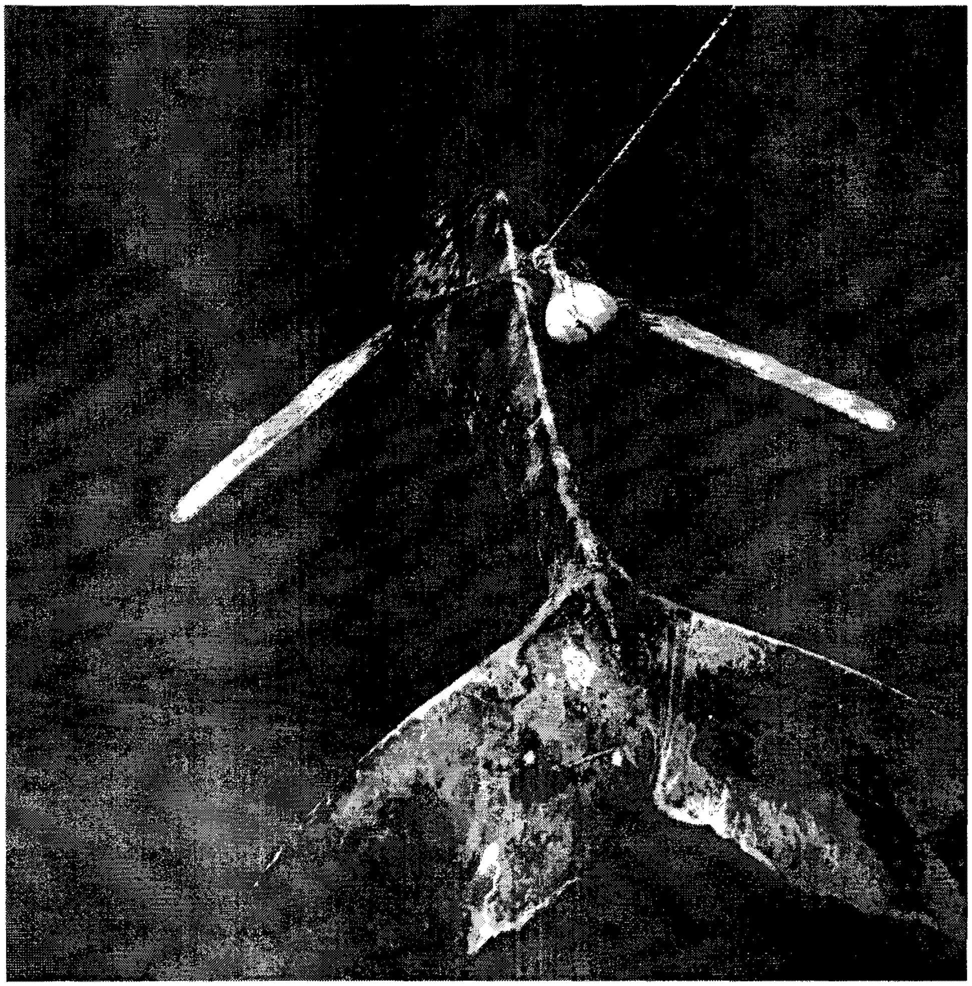

Figure 5. A humpback whale entangled in fisheries gear off Hawaii (Credit: NOAA, Hawaiian Islands Humpback Whale National Marine Sanctuary, ESA permit number 932-1489).

Several dozen individuals may become entangled and drown in commercial fishing gear (Atkinson et al. 2008; NMFS 2008g). Marine debris is also concerning for the health of Steller sea lion populations (Figure 6). It is estimated that 0.26% of Steller sea lions have marine debris around their necks or are hooked by fishing gear (0.07%); this equates to 100-200 individuals annually (FOC 2008; Raum-Suryan et al. 2009).

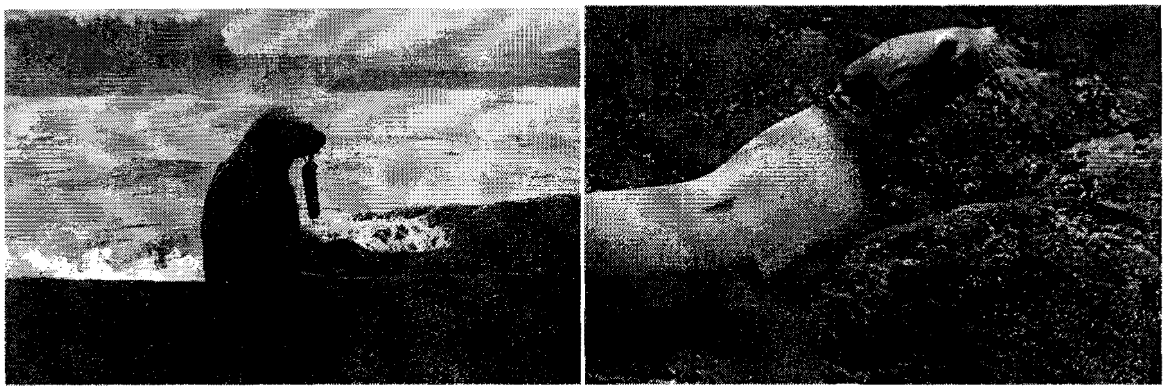

Figure 6. Steller sea lions having ingested a commercial fishing hook (A) and entangled by a rubber packing band. Photos taken and provided by Alaska Department of Fish and Game during research conducted under NOAA MMPA permits 14325 (A) and 358-1888 (B).

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 2008f; 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, and are 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 2008f).

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.

- 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. 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 in the action area 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 in the area 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.

Commercial harvest

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 the 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).

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 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 2002d; 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 2008f). 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 2008f; 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 2008f). In addition, private floatplanes, helicopters, and small aircraft regularly take advantage of whale watching opportunities (MMMP 2002b). Weather conditions in the Pacific Ocean in winter limit whale watching during this time and activity is greatest during summer (NMFS 2008f). 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 2008f).

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 2002d; 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). In addition to sound, the concentration of vessels surrounding killer whales in the region as raised concerns as to the amount of air pollution generated from vessel engines that individuals may be exposed to; Lachmuth et al. (2011) found that generally these pollutants are below levels which would cause adverse effects in humans, but this is not always the case. It is not know what effect, if any, air pollution may have on southern resident killer whales.

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. EEZ and have virtually no information on interactions between ships and commercial vessels outside of U.S. waters. With the information available, we know those interactions occur but we cannot estimate their significance to whale populations. Several humpback whales are also known to have become entangled in the North Pacific (Angliss and Outlaw 2007; Hill et al. 1997). Along the Pacific coast, a humpback whale is known to be killed about every other year by ship strikes (Barlow et al. 1997).

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)). 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 during this time. The vast majority of ship strike mortalities are never identified, and 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 (Waring et al. 2008), and a blue whale was struck and killed off the coast of California in 2002 (Jensen and Silber 2003b).

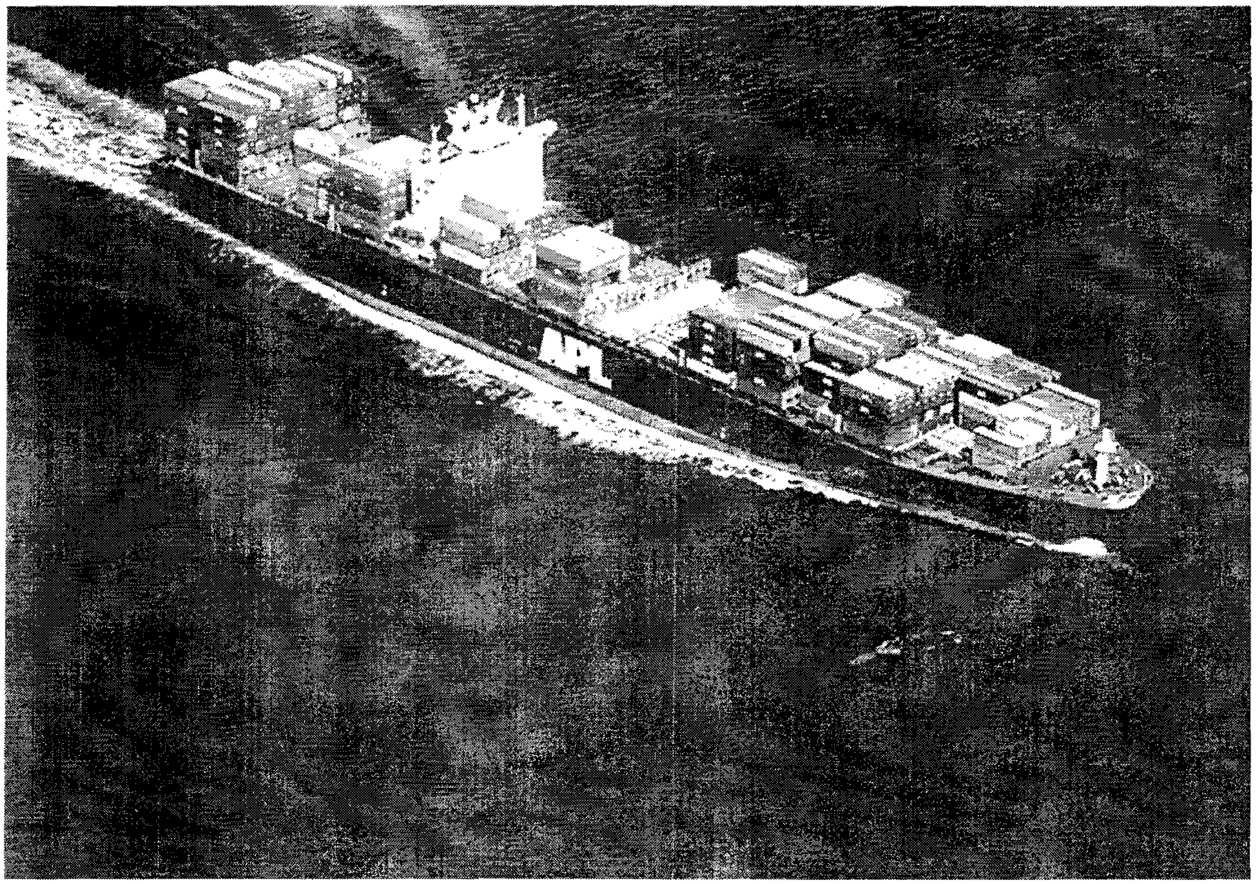

Figure 7. A near collision between a blue whale and a commercial cargo vessel in the Santa Barbara Channel Traffic Separation Scheme. Photo credit: NOAA Channel Islands National

Marine Sanctuary, 2002 (Permit CINMS-2002-001).

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 2009c). 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 2009c). 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).

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 Ocean, many of which extend into portions of the action area. Authorized research on ESA-listed whales includes close vessel and aerial approaches, biopsy sampling, breath 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 10-20 describe the cumulative number of takes for each listed species in the action area authorized in scientific research permits.

Table 10. Blue whale takes in the North Pacific.

Year	Approach	Biopsy	Suction cup tagging	Implantable tagging	Acoustic playback	Breath sampling
2009	13,044	1,065	313	105	0	0
2010	15,747	1,475	538	180	21	0
2011	23,677	2,570	3,455	1,295	21	1,020
2012	17,647	2,570	3,435	1,295	21	1,020
2013	12,902	2,460	3,295	1,250	21	1,020
Total	83,017	10,140	11,016	4,125	84	3,060

Permit numbers: 540-1811, 727-1915, 731-1774, 774-1714, 781-1824, 782-1719, 808-1735, 1058-1733, 1071-1770, 1127-1921, 14097, 14122, 14245, 14296, 14451, 14534, 14585, 15271, and 15330.

Table 11. Fin whale takes in the North Pacific.

Year	Approach	Biopsy	Suction cup tagging	Implantable tagging	Acoustic playback	Breath sample
2009	16,860	1,975	150	55	0	0
2010	21,283	2,975	410	130	80	0
2011	35,342	4,664	3,880	209	80	1,020
2012	24,642	4,664	3,880	209	80	1,020
2013	21,367	4,604	3,840	199	80	1,020
Total	119,494	18,882	12,160	802	320	3,060

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

Table 12. Humpback whale takes in the North Pacific.

Year	Approach	Biopsy	Suction cup tagging	Implantable tagging	Acoustic playback	Sampling sampling	Ultrasound
2009	44,399[1]	4,650	392	77	280	10	5
2010	67,171	6,060	1,447	237	970	10	5
2011	85,301	6,700	6,755	1,590	690	1,070	5
2012	50,697	6,500	6,585	1,565	990	1,070	5
2013	37,627	6,035	6,515	1,525	390	1,060	0
Total	285, 195	21,694	21,694	4,994	3,320	3,220	20

Permit numbers: 0642-1536, 0662-1661, 473-1700, 545-1761, 532-1822, 540-1811, 587-1767, 716-1705, 731-1774,753-1599, 727-1915, 774-1714, 781-1824, 782-1719, 808-1753, 945-1776, 965-1821, 1049-1718, 1058-1733, 1071-1770, 1120-1898, 1127-1921, 10018, 14097, 14122, 14245, 15271, 15274, 14296, 14353, 14451, 14534, 14585, 14599, 14610,14682, 13846, and 15330.

Table 13. 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	902	118	52	62	50
2011	1,302	188	62	132	50
2012	850	138	60	130	50
2013	703	138	24	94	50
Total	4,474	670	240	460	200

Permit numbers: 782-1719, 1058-1733, 13846, 14097, 14122, 14245, 14296, 14585, and 15330.

Table 14. Sei whale takes in the North Pacific.

Year	Approach	Biopsy	Suction cup tagging	Implantable tagging	Breath sampling
2009	4,170[1]	435	75	25	0
2010	6,396[1]	730	250	100	0
2011	7,871	638	548	115	1,060
2012	5,551	638	548	115	1,060
2013	5,331	628	558	115	1,060
Total	29,519	3,069	1,969	470	3,180

Permit numbers: 0642-1536, 540-1811, 727-1915, 731-1774, 774-1714, 782-1719, 808-1735, 1049-1718, 1058-1733, 1127-1921, 14097, 14122, 14245, 14296, 14451, 14534, 14585, and 15330.

Table 15. Southern resident killer whale takes in the North Pacific.

Year	Approach	Biopsy	Suction cup tagging	Breath 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 16. Sperm whale takes in the North Pacific.

Year	Approach	Biopsy	Suction cup tagging	Implantable tagging	Acoustic playback	Breath sampling
2009	17,895	770	100	40	0	0
2010	22,001	1,425	405	170	120	120
2011	34,621	3,785	2,885	380	120	1,060
2012	19,486	3,285	2,855	370	120	1,060
2013	18,476	3,165	2,855	360	120	1,060
Total	112,479	12,430	9,100	1,320	480	3,300

Permit numbers: 0642-1536, 473-1700, 540-1811, 727-1915, 731-1774, 774-1714, 781-1824, 782-1719, 1049-1718, 1071-1770,1127-1921, 14097, 14122, 14245, 14296, 14451, 14534, 14585, and 15330.

Table 17. 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	307,510	1,446	980	300	184	1,446	1,350	360	370	840	30
2010	273,261	1,446	980	300	181	1,446	1,350	360	370	840	30
2011	250,261	1,446	980	300	181	1,446	1,350	360	370	840	30
2012	248,461	1,446	980	300	181	1,446	1,350	360	370	840	30
2013	247,961	1,446	980	300	181	1,446	1,350	360	370	840	30
Total	1,327,454	7,230	4,900	1,500	908	7,230	6,750	1,800	1,850	4,200	150

Permit numbers: 532-1822, 540-1811, 715-1885, 731-1774, 774-1714, 965-1821, 1410-4M, 13430, 14097, 14245, 14325, 14326, 14336, 13846, 14337, 15330, 15483.

Table 18. Bocaccio takes in the North Pacific.

Year	Capture/handle	Mortality
2009	19	19
2010	46	46
2011	76	64
2012	25	13
2013	25	13
Total	191	155

Permits: 1586-3R, 14438, 14442, 14462, 15119, 15275, 15408, 15449, 15595, 16208, 16269, 16328, 16408,

and 16414.

Table 19. Yelloweye rockfish takes in the North Pacific.

Year	Capture/handle	Anesthesia	Mortality
2009	47	0	42
2010	163	0	139
2011	200	20	190
2012	23	0	13
2013	23	0	13
Total	456	20	397

Permits: 1586-3R, 14204, 14462, 14477, 15119, 15119-2A, 15225, 15431, 15449, and 15595.

Table 20. Canary rockfish takes in the North Pacific.

Year	Capture/handling	Anesthetize	Floy tag	Sonic tag	Mortality
2009	626	0	0	0	609
2010	682	0	15	15	615
2011	1,714	50	15	15	1,669
2012	25	0	0	0	13
2013	25	0	0	0	13
Total	3,072	50	30	30	2,919

Permits: 1586-3R, 14204, 14266, 14274, 14438, 14462, 14477, 14591, 14616, 15449, 15119, 15119-2A, 15225, 15237, 15269, 15275, 15408, 15431, 15476, 15545, 15595, 15847, 15849, 16017, 16166, 16208, 16220, 16269, 16271, 16292, 16327, 16328, 16408, 16414, 16429, and 16347.

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 permits 15569, 16111, 16160, and16163 would authorize "takes" by harassment of marine mammals during the proposed research by the applicant by directed and unintentional approach, satellite and sensory tagging (invasive and suction cup based), biopsy, breath sampling, acoustic playback, 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 or proposed 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 or proposed 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"[3] 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 whether 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 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 or proposed whales by research vessels
4. application of telemetry tags
5. continued attachment of tags
6. playback of simulated pile driving sounds, white noise, and ocean noise
7. fish- or whale-finding sonar

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

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 applicants propose to conduct aerial surveys, vessel surveys, and/or tagging of cetaceans in waters of the Pacific Ocean that would target numerous cetacean species, including listed blue, fin, sei, humpback, 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 (as well as Steller sea lions) could unintentionally be exposed to stressors associated with the proposed action, such as the potential for ship strike 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 ship strike to listed marine mammals under the proposed actions for these reasons. We therefore discount the potential for ship strike in association with the proposed actions to target species. We also discount the possibility that research activities would incidentally disturb listed species or species proposed for listing for the same reasons, except in cases where target individuals have conspecifics near them.

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 marine mammals.

Aerial surveys proposed by the applicant could co-occur with haul-out areas of some listed pinnipeds, including the eastern DPS of Steller sea lions. These species may respond to the sight or sound of the aircraft, possibly causing them to temporarily abandon their haul-out or change their behavioral state. Therefore, unintentional harassment of these listed or proposed for listing species is possible. Guadalupe fur seals take requests have been included by the Permit's Division due to the species occurrence (and possible exposure to vessel surveys) in southern California. However, the species' occurrence here is exceptionally rare and we do not expect exposure to surveys.

Accordingly, this consultation focused on the following stressors likely to occur from the proposed activities and may adversely affect ESA-listed species: Overflights of marine mammals by survey aircraft; close approaches to whales and Steller sea lions by research vessels; application and continued attachment of tags; continued attachment of tags; playback of simulated pile driving sounds, white noise, and ocean noise; and fish- or whale-finding sonar.

Exposure analysis

Exposure analyses identify the ESA-listed or proposed species that are likely to co-occur with the actions' effects on the environment in space and time, and identify the nature of that co-occurrence. The *Exposure analysis* identifies, as possible, the number, age or life stage, and gender of the individuals likely to be exposed to the actions' effects and the population(s) or subpopulation(s) those individuals represent.

Our exposure analysis began with identifying the stressors that listed resources are likely to be exposed to, which we did in the preceding section. We continue by identifying the amount or extent of exposure that we believe is reasonably likely to occur. The Permit's Division provided estimated take numbers (Tables 21-24) in their initiation package for each of the four permits included in this consultation. However, these estimates were not supported by a clear, reproducible methodology. We therefore adopted an alternative methodology that relied upon past performance to assess likely exposure for some actions, including approaches and tagging for which significant data existed from the applicant's previous research that mirrored the proposed research. This analysis incorporated likely additional future exposure where the Permit's Division could provide sufficient detail; the numerical output of this is outlined in Tables 25-28. Furthermore, additional qualitative assessment was incorporated for listed resources that had not been documented to be previously exposed by the applicants, but which may reasonably be exposed in the future, or where we did not feel enough past performance data were available to rely solely upon for future estimates. We performed a binomial distribution on the likely exposure calculated in Tables 25-28 to identify the expected range of exposures per individual (for those species which are likely to be exposed) as well as the most probable number of exposures.

The proposed actions are to issue several permits authorizing suction-cup tagging, satellite tagging, close approaches, photo-ID, acoustic recording, exhalation sampling, and behavioral observations of marine mammals. These actions and the number of "takes" proposed to be authorized by the Permits Division are outlined in Tables 21-24.

Table 21. Number of animals proposed to be taken by species, life stage, and action under Permit 15569.

Species-population	**Life stage**	**Proposed number of individuals taken annually**	**Action**
Blue whale (*Balaenoptera musculus*)-North Pacific	All	100	Close approach-aerial or vessel
Fin whale (*Balaenoptera physalus*)- North Pacific	All	100	Close approach-aerial or vessel
Sei whale (*Balaenoptera borealis*)-North Pacific	All	100	Close approach-aerial or vessel
Humpback whale (*Megaptera novaeangliae*)-North Pacific	All	300	Close approach-aerial or vessel
North Pacific right whale (*Eubalaena japonica*)-North Pacific	All	10	Close approach-aerial or vessel
Sperm whale (*Physeter macrocephalus*)-North Pacific	All	100	Close approach-aerial or vessel

Species-population	Life stage	Proposed number of individuals taken annually	Action
Southern resident killer whale (*Orcinus orca*)-Southern resident	All	8,500	Close approach-aerial or vessel
Steller sea lion (*Eumetopias jubatus*)-Eastern or western DPS	All	100	Close approach-aerial or vessel

Table 22. Number of animals proposed to be taken by species, life stage, and action under Permit 16111.

Species-population	Life stage	Proposed number of individuals taken annually	Action
Blue whale (*Balaenoptera musculus*)-North Pacific	>4 months old	100	Close approach-vessel, prey mapping, exhalation sample, passive recording, biopsy, underwater photography
	>4 months old	80	Close approach-vessel, prey mapping, exhalation sample, passive recording, biopsy, underwater photography, suction cup tagging, dart tagging (up to 50 suction, up to 30 dart, up to 6 suction and dart tagged)
	All	2,000	Close approach-aerial or vessel, prey mapping, exhalation sample, passive recording, underwater photography
Fin whale (*Balaenoptera physalus*)- North Pacific	>4 months old	100	Close approach-vessel, prey mapping, exhalation sample, passive recording, biopsy, underwater photography
		60	Close approach-vessel, prey mapping, exhalation sample, passive recording, biopsy, underwater photography, suction cup tagging, dart tagging (up to 30 suction, up to 30 dart, up to 6 suction and dart tagged)
	All	1,500	Close approach-aerial or vessel, prey mapping, exhalation sample, passive recording, underwater photography
Sei whale (*Balaenoptera borealis*)-North	>1 year old	20	Close approach-vessel, prey mapping, exhalation sample, passive recording, biopsy, underwater

Species-population	**Life stage**	**Proposed number of individuals taken annually**	**Action**
Pacific			photography
		15	Close approach-vessel, prey mapping, exhalation sample, passive recording, biopsy, underwater photography, suction cup tagging, dart tagging (up to 5 suction, up to 10 dart, up to 1 suction and dart tagged)
		70	Close approach-aerial or vessel, prey mapping, exhalation sample, passive recording, underwater photography
Humpback whale (*Megaptera novaeangliae*)-North Pacific	>4 months old	100	Close approach-vessel, prey mapping, exhalation sample, passive recording, biopsy, underwater photography
		80	Close approach-vessel, prey mapping, exhalation sample, passive recording, biopsy, underwater photography, suction cup tagging, dart tagging (up to 50 suction, up to 30 dart, up to 6 suction and dart tagged)
	All	2,000	Close approach-aerial or vessel, prey mapping, exhalation sample, passive recording, underwater photography
Sperm whale (*Physeter macrocephalus*)-North Pacific	>1 year old	100	Close approach-vessel, prey mapping, exhalation sample, passive recording, biopsy, underwater photography
		60	Close approach-vessel, prey mapping, exhalation sample, passive recording, biopsy, underwater photography, suction cup tagging, dart tagging (up to 40 suction, up to 20 dart, up to 4 suction and dart tagged)
	All	700	Close approach-aerial or vessel, prey mapping, exhalation sample, passive recording, underwater photography
Southern resident killer whale (*Orcinus orca*)-Southern resident	All	300	Close approach-aerial or vessel

Species-population	Life stage	Proposed number of individuals taken annually	Action
Steller sea lion (*Eumetopias jubatus*)-Eastern DPS	All	500	Incidental disturbance via close approach-vessel or ground monitoring
		100	Incidental disturbance via close approach-vessel

Table 23. Number of animals proposed to be taken by species, life stage, and action under Permit 16160.

Species-population	Life stage	Proposed number of individuals taken annually	Action
Humpback whale (*Megaptera novaeangliae*)-North Pacific	All	10	Close approach-vessel
Southern resident killer whale (*Orcinus orca*)-Southern resident	All	200	Close approach-vessel

Table 24. Number of animals proposed to be taken by species, life stage, and action under Permit 16163.

Species-population	Life stage	Proposed number of individuals taken annually	Action
Blue whale (*Balaenoptera musculus*)-North Pacific	All	200	Close approach-aerial
		200	Close approach-vessel, prey mapping, passive recording
	Juvenile/adult	30	Close approach-vessel, prey mapping, passive recording, biopsy
		20	Close approach-vessel, prey mapping, passive recording, biopsy, suction cup tagging, dart tagging (up to 10 suction, up to 10 dart, up to 10 suction and dart tagged)
Fin whale (*Balaenoptera physalus*)- North	All	500	Close approach-aerial
		500	Close approach-vessel, prey mapping, passive recording

Species-population	**Life stage**	**Proposed number of individuals taken annually**	**Action**
Pacific	Juvenile/adult	30	Close approach-vessel, prey mapping, passive recording, biopsy
		30	Close approach-vessel, prey mapping, passive recording, biopsy, suction cup tagging, dart tagging (up to 10 suction, up to 20 dart, up to 10 suction and dart tagged)
		5	Close approach-vessel, acoustic playback, prey mapping
Sei whale (*Balaenoptera borealis*)-North Pacific	All	500	Close approach-aerial
		500	Close approach-vessel
Humpback whale (*Megaptera novaeangliae*)-North Pacific	All	500	Close approach-aerial
		500	Close approach-vessel, prey mapping, passive recording
		5	Close approach-vessel, acoustic playback, prey mapping
	Juvenile/adult	30	Close approach-vessel, prey mapping, passive recording, biopsy
		10	Close approach-vessel, prey mapping, passive recording, exhalation sampling
		20	Close approach-vessel, prey mapping, passive recording, biopsy, suction cup tagging, dart tagging (up to 10 suction, up to 10 dart, up to 10 suction and dart tagged)
North Pacific right whale (*Eubalaena japonica*)-North Pacific	All	10	Close approach-aerial
		10	Close approach-vessel
Unidentified baleen whale	All	500	Close approach-vessel
		500	Close approach-aerial

Species-population	**Life stage**	**Proposed number of individuals taken annually**	**Action**
Sperm whale (*Physeter macrocephalus*)-North Pacific	All	250	Close approach-aerial
		250	Close approach-vessel, prey mapping, passive recording
	Juvenile/adult	30	Close approach-vessel, prey mapping, passive recording, biopsy
		20	Close approach-vessel, prey mapping, passive recording, biopsy, suction cup tagging, dart tagging (up to 10 suction, up to 10 dart, up to 10 suction and dart tagged)
False killer whale (*Pseudorca crassidens*)-Hawaiian insular	All	200	Close approach-vessel, prey mapping, passive recording, exhalation sampling
	Juvenile/ adult	20	Close approach-vessel, prey mapping, passive recording, biopsy, suction cup tagging, dart tagging
Southern resident killer whale (*Orcinus orca*)-Southern resident	All	6,000	Close approach-aerial
		6,000	Close approach-vessel, prey mapping, passive recording
		750	Close approach-vessel, acoustic playback, prey mapping, passive recording
		130	Incidental-close approach-vessel, acoustic playback, prey mapping, passive recording
	Juvenile/ adult	50	Close approach-vessel, prey mapping, passive recording, exhalation sampling
		39	Close approach-vessel, prey mapping, passive recording, biopsy, suction cup tagging, dart tagging (up to 30 suction, up to 9 dart, up to 9 suction and dart tagged)
		25	Close approach-vessel, prey mapping, passive recording, biopsy

Species-population	Life stage	Proposed number of individuals taken annually	Action
		25	Close approach-vessel, prey mapping, passive recording, exhalation sampling, ultrasound
Steller sea lion (*Eumetopias jubatus*)-Eastern DPS	All	500	Close approach-aerial
		500	Incidental close approach-vessel
		5	Incidental close approach-vessel, acoustic playback, prey mapping
Guadalupe fur seal (*Arctocephalus townsendi*)	All	100	Close approach-aerial
Hawaiian monk seal (*Monachus schauinslandi*)	All	10	Incidental close approach-vessel

The applicants expect that an individual of any age class or sex may be approached numerous times per year under the proposed activities (maximums: most baleen whales = 10; North Pacific right whale = 2; Steller sea lions = 5; Guadalupe fur seals = 3; Hawaiian monk seals = 2; sperm whales = 10; southern resident killer whales = 100; and Hawaiian insular false killer whales = 3). No neonates would be allowed to be tagged but juveniles, subadult, and adult age classes of any sex may be tagged with two tags (one skin penetrating and one suction-attached) simultaneously under permits 16111 and 16163. The permits would be conditioned to minimize harassment from tagging activities to no more than two times per day and no more than three times per day for biopsy attempts. However, the applicants generally expect that tagging would be successful within two attempts for each activity; monitoring reports indicate this is reasonable. No individual may be tagged with both suction cup and implantable tags more than once per year. This difference is based upon limitations researchers have requested in their separate permit applications. Tagging attempts must be discontinued if repetitive strong reactions are found.

Although these activities are proposed to occur, we expect the level of exposure to these activities for most listed species to be different than the levels of "take" requested above (Tables 25-28). This is based upon annual monitoring reports of the applicant's activities that include activities similar or identical to those in the proposed permits. 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. In addition, we assume 4% annual population growth for all populations that are currently increasing in abundance or stable. These values contributed to Tables 25-28.

For species and action combinations which we did not have prior exposure data from researchers prior monitoring reports, we determined whether this was due to low or no effort, or if the requested take represented a new or expanded activity. In no case did we receive information indicating any applicant was significantly expanding prior activities or undertaking the same activities into new areas. With the exception of dart tagging southern resident killer whales, active acoustic playback, and ultrasound work under permit 16163, we also did not receive information to suggest applicants intended to undertake new activities. The Permit's Division did inform us that applicant activities over the five year duration of the proposed permits would be dependent upon funding, among other variables, that are unpredictable and poorly-known at present. We acknowledge this and feel this variability is adequately represented in the amount of effort (and consequent exposure) that is documented in the monitoring reports provided by the applicants and Permit's Division. When data were lacking to quantify a likely level of exposure based upon past researcher performance, we qualitatively assessed the level of reasonably likely exposure. We qualitatively assigned reasonably likely levels of exposure based upon expected researcher effort and co-occurrence of listed species with this effort. We assumed no significant change in effort is expected and the past level of effort did not result in sizeable levels of exposure. We acknowledge that although exposure to these species is not expected to be frequent, if exposure would occur it can occur to numerous individuals. For example, if researchers encountered sperm whales, it is highly probable that a family group may be encountered, involving several individuals. Other species, such as sei whales, rarely occur in groups, but have been documented to occur in concentrated numbers during times of high prey availability in limited areas. Under such conditions, numerous individuals may be exposed to researcher activities. To account for these circumstances, we include considerations of group size in the region and occurrence patterns (as revealed by sighting histories from regional surveys) in our qualitative estimates of exposure for listed species to actions under the proposed permits.

Table 25. Expected annual exposure events for listed species to proposed activities under permit 15569.

Species-population	**Expected annual exposure**	**Activity**
Blue whale (*Balaenoptera musculus*)-North Pacific	3[1]	Close approach-vessel
Fin whale (*Balaenoptera physalus*)-North Pacific	3[1]	Close approach-vessel
Sei whale (*Balaenoptera borealis*)-North Pacific	6[1]	Close approach-vessel
Humpback whale (*Megaptera novaeangliae*)-North Pacific	4[1]	Close approach-vessel
North Pacific right whale (*Eubalaena japonica*)-North Pacific	1[1]	Close approach-vessel
Sperm whale (*Physeter macrocephalus*)-North Pacific	30[1]	Close approach-vessel

Species-population	Expected annual exposure	Activity
Southern resident killer whale (*Orcinus orca*)-Southern resident	3,960	Close approach-vessel
Steller sea lion (*Eumetopias jubatus*)-Eastern DPS	100[2]	Unintentional harassment via close approach-vessel

[1] Best estimate of average group size in the region (rounded to the next whole number) to reflect a single possible encounter.

[2] Adopted from applicant's estimate of potential exposure. Considering the Permit's Division recent requirement of applicant's take reporting to be more reflective of exposure versus response (as it previously was), we expect future monitoring reports to be more informative of true exposure. We presently have no better method to determine likely exposure for Steller sea lion exposure than the applicant's estimate based upon his/her previous research activities that are very similar to those in the proposed permit.

Expected exposure numbers derived from past performance data were rounded to the next highest multiple of 10 to reflect the analytical uncertainty in all vessel, tagging, and biopsy activities on all species. The exception to this was North Pacific right whales, whose rarity makes exposure to close vessel approach and tagging unlikely to be more than one instance annually under a given permit.

Table 26. Expected annual exposure events for listed species to proposed activities under permit 16111.

Species-population	Expected annual exposure	Activity
Blue whale (*Balaenoptera musculus*)-North Pacific	1,380	Close approach-aerial or vessel*
	110[1]	Tagging
	140	Biopsy
Fin whale (*Balaenoptera physalus*)- North Pacific	250	Close approach-aerial or vessel*
	50	Tagging
	80	Biopsy
Sei whale (*Balaenoptera borealis*)-North Pacific	6	Close approach-aerial or vessel*
	2[2]	Tagging
	2[2]	Biopsy
Humpback whale (*Megaptera*	1,500	Close approach-aerial or vessel*

Species-population	Expected annual exposure	Activity
novaeangliae)-North Pacific	60	Tagging
	220	Biopsy
Sperm whale (*Physeter macrocephalus*)-North Pacific	90	Close approach-aerial or vessel*
	30	Tagging
	10	Biopsy
Southern resident killer whale (*Orcinus orca*)-Southern resident	340[3]	Close approach-aerial or vessel
Steller sea lion (*Eumetopias jubatus*)-Eastern DPS	130	Unintentional harassment via close approach-aerial

*We assume breath sampling and/or passive acoustic recording can occur on any close approach. These additional activities are not expected to make a given exposure more or less significant to any listed individual.

1 Permit would limit suction cup tagging to 50 and dart tagging to 30, or 80 combined tagging attempts

2 No data were available to inform the probability of tagging or biopsying sei whales. We determined a reasonably likely value by evaluating the maximum tag/biopsy rates of other large whale species by the applicant and found up to one-third of encountered individuals may be exposed to either tagging or biopsying. This proportion was applied to expected sei whale tagging and biopsying attempts.

3 Permit would limit approaches to 300 individuals.

Table 27. Expected annual exposure events for listed species to proposed activities under permit 16160.

Species-population	Expected annual exposure	Activity
Humpback whale (*Megaptera novaeangliae*)-North Pacific	4[1]	Close approach-vessel
Southern resident killer whale (*Orcinus orca*)-Southern resident	260[2]	Close approach-vessel

[1] Best estimate of average group size in the region (rounded to the next whole number) to reflect a single possible encounter.

[2] Permit would limit approaches to 200.

Table 28. Expected annual exposure events for listed species to proposed activities under permit 16163.

Species-population	Expected annual exposure	Activity
Blue whale (*Balaenoptera musculus*)-North Pacific	3[1]	Close approach-aerial or vessel*
	3[2]	Tagging
	3[2]	Biopsy
Fin whale (*Balaenoptera physalus*)- North Pacific	3[1]	Close approach-vessel*
	3[2]	Tagging
	3[2]	Biopsy
	3[2]	Active playback
Sei whale (*Balaenoptera borealis*)-North Pacific	6[1]	Close approach-vessel*
Humpback whale (*Megaptera novaeangliae*)-North Pacific	4[1]	Close approach-vessel*
	4[2]	Tagging
	4[2]	Biopsy
North Pacific right whale (*Eubalaena japonica*)-North Pacific	1[1]	Close approach-aerial or vessel*
Unidentified baleen whale	1	Close approach-aerial or vessel
Sperm whale (*Physeter macrocephalus*)-North Pacific	30[1]	Close approach-vessel*
	30[2, 3]	Tagging
	30[2]	Biopsy
Southern resident killer whale (*Orcinus orca*)-Southern resident	1,430	Close approach-vessel*
	10	Tagging
	30[4]	Biopsy
	880[5]	Active playback

Species-population	Expected annual exposure	Activity
	25[5]	Ultrasound
False killer whale (*Pseudorca crassidens*)-Hawaiian insular	30[1]	Close approach-vessel
	30[2]	Tagging
Steller sea lion (*Eumetopias jubatus*)-Eastern DPS	1,000[6]	Unintentional harassment via close approach-aerial or vessel
	5[5]	Unintentional harassment via acoustic playback

*We assume breath sampling and/or passive acoustic recording can occur on any close approach. These additional activities are not expected to make a given exposure more or less significant to any listed individual.

[1] Best estimate of average group size in the region (rounded to the next whole number) to reflect a single possible encounter.

[2] No past performance data were available to quantitatively estimate exposure. Also unavailable were data to indicate what proportion of encountered individuals of other similar species the research has tagged or biopsied. We provisionally assume all individuals encountered may be exposed to authorized activities pending more informative data.

[3] Permit limits tagging to 10 suction cup and 10 dart tagging attempts.

[4] Permit limits biopsy to 25 attempts.

[5] New activity undertaken by applicant. We could not identify a similar activity on this or a surrogate species and therefore provisionally accept the applicant's estimate pending additional exposure data.

[6] Adopted from applicant's estimate of potential exposure. Considering the Permit's Division recent requirement of applicant's take reporting to be more reflective of exposure versus response (as it previously was), we expect future monitoring reports to be more informative of true exposure. We presently have no better method to determine likely exposure for Steller sea lion exposure than the applicant's estimate based upon his/her previous research activities that are very similar to those in the proposed permit.

Although the Permit's Division estimated the number of exposures to individuals, researchers cannot determine one individual from another in the field (with the exception of southern resident killer whales and Hawaiian insular false killer whales after they have been approached and photoidentified). Therefore, we used a binomial 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 (Tables 29-232). This represents the best available method to estimate the number of exposures that an individual is likely to receive for the actions and circumstances under consideration. We qualitatively expect that no Steller sea lion would be exposed to any action more than once annually.

Table 29. Expected exposures of proposed activities to individuals annually and over permit 15569's duration.

Species/activity	Annual		Over permit's lifespan	
	Most probable individual exposure[1]	Range of individual exposures	Most probable individual exposure[1]	Range of individual exposures
Blue whale (*Balaenoptera musculus*)-North Pacific				
Close approach-vessel	1	0-1	1	0-1
Fin whale (*Balaenoptera physalus*)- North Pacific				
Close approach-vessel	1	0-1	1	0-1
Sei whale (*Balaenoptera borealis*)-North Pacific				
Close approach-vessel	1	0-1	1	0-1
Humpback whale (*Megaptera novaeangliae*)-North Pacific				
Close approach-vessel	1	0-1	1	0-1
North Pacific right whale (*Eubalaena japonica*)-North Pacific				
Close approach-vessel	1	0-1	1	0-1
Sperm whale (*Physeter macrocephalus*)-North Pacific				
Close approach-vessel	1	0-1	1	0-2
Southern resident killer whale (*Orcinus orca*)				
Close approach-vessel	44	23-69	222	176-273

[1]Assusming an exposure actually occurs, minimum value = 1

Table 30. Expected exposures of proposed activities to individuals annually and over permit 16111's duration.

Species/activity	Annual		Over permit's lifespan	
	Most probable individual exposure[1]	Range of individual exposures	Most probable individual exposure[1]	Range of individual exposures
Blue whale (*Balaenoptera musculus*)-North Pacific				
Vessel approach	1	0-4	1	0-7
Tagging	1	0-2	1	0-2
Biopsy	1	0-2	1	0-3
Fin whale (*Balaenoptera physalus*)- North Pacific				
Vessel approach	1	0-2	1	0-2
Tagging	1	0-1	1	0-2
Biopsy	1	0-1	1	0-2
Sei whale (*Balaenoptera borealis*)-North Pacific				
Vessel approach	1	0-1	1	0-1
Tagging	1	0-1	1	0-1
Biopsy	1	0-1	1	0-1
Humpback whale (*Megaptera novaeangliae*)-North Pacific				
Vessel approach	1	0-2	1	0-4
Tagging	1	0-1	1	0-2

Species/activity	Annual		Over permit's lifespan	
	Most probable individual exposure[1]	Range of individual exposures	Most probable individual exposure[1]	Range of individual exposures
Biopsy	1	0-1	1	0-2
Sperm whale (*Physeter macrocephalus*)-North Pacific				
Vessel approach	1	0-1	1	0-2
Tagging	1	0-1	1	0-2
Biopsy	1	0-1	1	0-1
Southern resident killer whale (*Orcinus orca*)				
Vessel approach	3	0-12	16	4-33

[1] Assusming an exposure actually occurs, minimum value = 1

Table 31. Expected exposures of proposed activities to individuals annually and over permit 16160's duration.

Species/activity	Annual		Over permit's lifespan	
	Most probable individual exposure[1]	Range of individual exposures	Most probable individual exposure[1]	Range of individual exposures
Humpback whale (*Megaptera novaeangliae*)-North Pacific				
Vessel approach	1	0-1	1	0-1
Southern resident killer whale (*Orcinus orca*)				
Vessel approach	2	0-9	11	0-25

[1] Assusming an exposure actually occurs, minimum value = 1

Table 32. Expected exposures of proposed activities to individuals annually and over permit 16163's duration.

Species/activity	Annual		Over permit's lifespan	
	Most probable individual exposure[1]	Range of individual exposures	Most probable individual exposure[1]	Range of individual exposures
Blue whale (*Balaenoptera musculus*)-North Pacific				
Aerial/vessel approach	1	0-1	1	0-1
Tagging	1	0-1	1	0-1
Biopsy	1	0-1	1	0-1
Fin whale (*Balaenoptera physalus*)- North Pacific				
Vessel approach	1	0-1	1	0-1
Tagging	1	0-1	1	0-1
Biopsy	1	0-1	1	0-1
Acoustic playback	1	0-1	1	0-1
Sei whale (*Balaenoptera borealis*)-North Pacific				
Vessel approach	1	0-1	1	0-1

Species/activity	Annual		Over permit's lifespan	
	Most probable individual exposure[1]	Range of individual exposures	Most probable individual exposure[1]	Range of individual exposures
Humpback whale (*Megaptera novaeangliae*)-North Pacific				
Vessel approach	1	0-1	1	0-1
Tagging	1	0-1	1	0-1
Biopsy	1	0-1	1	0-1
North Pacific right whale (*Eubalaena japonica*)-North Pacific				
Aerial/vessel approach	1	0-1	1	0-1
Sperm whale (*Physeter macrocephalus*)-North Pacific				
Vessel approach	1	0-1	1	0-2
Tagging	1	0-1	1	0-1
Biopsy	1	0-1	1	0-2
Unidentified baleen whale				
Aerial/vessel approach	1	0-1	1	0-1
Southern resident killer whale (*Orcinus orca*)				
Vessel approach	16	4-32	80	51-112
Tagging	1	0-4	1	0-7
Biopsy	1	0-3	1	0-5
Active playback[2]	9	1-23	49	27-75
Ultrasound	1	0-4	1	0-7
False killer whale (*Pseudorca crassidens*)-Hawaiian insular				
Aerial/vessel approach	1	0-3	1	0-6
Tagging	1	0-3	1	0-6

[1] Assusming an exposure actually occurs, minimum value = 1

[2] The ability of researchers to discriminate individuals prior to undertaking this activity means that permit-imposed conditions will likely be effective in limiting exposure to five active acoustic playbacks per year.

Response analysis

As discussed in the *Approach to the assessment* section of this Opinion, response analyses determine how listed or proposed 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 1998; Cowan and Curry 2002; Cowan and Curry 2008; Herraez et al. 2007). The most widely-recognized indicator of vertebrate stress, cortisol, normally takes hours to days to return to baseline levels following a significantly stressful event, but other hormones of the 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).

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 applicant), 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.

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. Ljungblad (1981) did not observe responses to the survey aircraft while observing seismic surveys were occurring nearby. 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 or below 182 m and less than 250 m laterally from target individuals (Patenaude et al. 2002). Payne et al. (1983) found that another balaenid, southern right whales, rarely reacted 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.

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 helicopter was closer than 800 m away. Richardson et al. (1985a) reported that >1,000 animals stampeded off a beach in response to a Bell 205 helicopter greater than 1.6 kilometers away. Data from the National Marine Mammal Laboratory (NMML), which routinely conducts aerial surveys for both cetaceans and pinnipeds, has found that pinnipeds may be disturbed by entering the water when overflown. Pinnipeds may also or alternatively lift their heads to observe the aircraft. The NMML has found that <10% of Steller sea lions respond to aircraft overflights, with individuals in water responding at an even lower frequency.

Of the applicants consulting here, only Dr. Balcomb has conducted aerial surveys in the Pacific Ocean under their current permits, although permit 16163 would also allow for continued authorization of listed marine mammals via aerial survey. Data from the NMML, which has conducted much more extensive aerial survey effort in the North Pacific has not documented responses of several cetacean species seen during surveys, including fin, sei, humpback, and North Pacific right whales. Expert opinion further bolsters this position (Laura Morse, NMFS, pers. comm.; Trisia Naessig, pers. comm.). Based upon the lack of response in the NMML's more extensive surveys and the anticipated brevity of effort by the applicants (if it occurs at all), we do not expect any individual blue, fin, sei, humpback, or North Pacific right to respond to survey planes. However, the applicant may deviate from methods used by NMML (such as the use of helicopters instead of NMML's exclusive use of small planes), which may change the response rate of target individuals. Therefore, it is possible that a few individuals of these species may respond to overflights with startle responses, rapid dives, or changes in direction. We expect the same response type and frequency for blue and sperm whales, with sperm whales also possibly delaying click production during dives or forming a group defensive posture. We do not anticipate any individual of these species will be re-exposed due to the wide-ranging nature of these taxa and likely low survey effort.

We also expect a few individual southern resident killer whales may respond to aircraft overflights with startle responses, rapid dives, or changes in direction. Due to the more restricted ranges of these species, re-exposure may occur; however, assuming individuals experience re-exposure, we expect the same responses will not necessarily occur with every re-exposure and will vary by individual and context. Some would likely be the same and some more or less pronounced.

Aerial surveys may expose Steller sea lions (surveys along the U.S. west coast) to this activity. Given the lack of aerial survey history by the applicants and the low proportion of exposed individuals responding to aerial survey overflights, we expect a few individuals Steller sea lions may respond to aerial survey activities with head lifts or dives.

Close approaches-surface vessel

Vessel approaches have the potential to induce behavioral and physiological changes in targeted individuals. 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. 2001b; 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 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 to 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. The *Environmental baseline* identified changes in southern resident killer whale vocal patterns that are thought to be a result of long-term exposure to high levels of vessel traffic.

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.

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 2002d; Foote et al. 2004b; Holt et al. 2011; 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 applicants propose to use one vessel per survey (except for permit 15569, which may involve two vessels simultaneously), 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 before the vessel breaks contact following photoidentification, acoustic recording, tagging, exhalation sampling, and/or behavioral documentation.

No quantitative assessment of Steller sea lion disturbance by approaching vessels is available, as significant variation exists (0-100% of individuals responding). Eastern DPS Steller sea lions are more amendable to approach than western Steller sea lions and less likely to flush (Brian Fadely, NMFS-NMML, pers. comm.). Although we cannot determine a likely number of individuals responding, we expect any response by Steller sea lions to consist of ephemeral alerting and possible barking at the research vessel. No significant biological consequence is expected for any individual Steller sea lion.

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. Based upon the available literature and anticipated levels of future exposure, one to a few dozen blue, fin, sei, humpback, and sperm whales may also respond annually with low-to moderate-level behavioral responses described above. Although thousands of southern resident killer whale approaches may occur per year (and have during the applicants' previous activities), response to the applicant's activities has yet to be observed. However, it is possible that one or a few responses may occur to close vessel approaches (with low- to moderate-level behavioral responses). Tables 29-32 on pages 125-127 describes 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. Responses to cumulative approaches is addressed further under *Integration and synthesis*.

Tagging

Partially implantable and 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% 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 some whales to track movements and dive data. Implantable tags can cause behavioral responses similar to close approach as well as wounds, bruising, swelling, hydrodynamic drag. Some species are more behaviorally responsive than others, as shown in Table 33. Humpback whales tend to be one of the least responsive baleen whales to tagging, while sperm whales are highly responsive. 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.

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 applicant 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 titanium, and thoroughly disinfected before attachment. Most infections in wildlife resulting from invasive tagging stem from the skin (Hawkins 2004a; Mate et al. 2007b). Invasive components are generally designed to minimize the potential for skin intrusion into the wound at time of tagging (Mate et al. 2007b). 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. 2007b; 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. 2007b). 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). Divots are theorized to stem from fat cell rupture upon tag entry (Mate et al. 2007b). 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 in tagged whales.

Table 33. Number of whales tagged and number of whales responding to the tagging process, by species, including failed attempts (Mate et al. 2007b).

Whale species	Number tagged	Number responding	Percent responding
Blue	146	22	15.1
Fin	29	13	44.8
Humpback	122	37	30.3
Sperm	60	51	85

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. 2007b). 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. 2007b). Only two studies of a wound after tagging. One was based upon a gray whale that stranded dead 18 days post tagging; although the animal 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). The other study was of a North Atlantic right whale into which a sedation dart was deep-penetrating sedation dart was inserted. Post mortem examination revealed extensive muscle tissue damage, likely as a result of the aforementioned shear forces between the blubber (which likely anchored the dart) and muscle (which being less dense, was "shredded" by the dart tip) (McLellan 2011).

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. 2007b). 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, 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).

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. 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. 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)).

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 (1978b) 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.

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

Response rate data from Mate et al. (2007b) lead us to believe that up to five blue whales would respond to dart tagging under permit 16111 and up to one individual would respond under permit 16163. These responses would include low to moderate behavioral responses. Localized swelling and an immune response are also expected.

Sei whales. We do not have specific information to inform us as to how many sei whales may respond to dart tagging. As we only expect two individuals to be exposed to this annually, we provisionally assume all individuals may respond to this stressor in manners similar to other balaenopterids (skin twitching, startle reactions or flinching, altered swimming speed and orientation, diving, rolling, head lifts, high back arching, fluking, and tail swishing). As with other species, we also expect localized swelling and an immune response are also expected.

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 (Grahl-Nielsen et al. 2010).

We expect that, of up to 30 and three fin whales exposed to dart tagging under permits 16111 and 16163, respectively, up to 14 and two individuals would respond, respectively, with startle, twitching, and /or fast dive responses, with individuals returning to baseline behavior within seconds or minutes (Mate et al. 2007b). Localized swelling and an immune response are also expected.

Humpback whales. Short-term, behavioral effects are also known 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. 2001b; 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. 2007b). "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). The applicant's monitoring data indicate that fast dive, tail flicks, and rolling are responses of humpback whales to tagging.

Response rate data from Mate et al. (2007b) lead us to believe that up to 10 humpback whales would respond to dart tagging under permit 16111 and up to two individuals would respond under permit 16163. These responses would include low to moderate responses such as skin twitching, startle reactions or flinching, altered swimming speed and orientation, diving, rolling, head lifts, high back arching, fluking, and tail swishing. Localized swelling and an immune response are also expected.

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. 2007b). 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).

Based upon response rate data from Mate et al. (2007b), we expect sperm whales to respond to dart tagging attempts under permit 16111 up to 17 times and to dart tagging under permit 16163 up to nine times. Responses would include ephemeral twitches, startles, and/or dives. Localized swelling and an immune response are also expected.

Killer whales. Although satellite dart tags have not previously been widely used on the southern resident killer whale DPS, similar tags have been recently deployed on other populations of killer whales, and the results of those studies inform this assessment. 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 2008b)). 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 in permit 16163. 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 2008b)). 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 2008b)).

Andrews et al. (2005b) 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 proposed in permit 16163, 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 (1978b) 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 satellite tags (40 g), 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 1978b). 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.

Because implantable tags would penetrate the skin of targeted killer whales, their use poses a risk due to infection at the tag site. We assume that southern resident killer whales might respond separately to the strike of the dart tag and infection at the point of penetration. Limited information exists on the potential for infection at tag sites, which has not been the subject of much focused study in whales. However, some information from observations of tagged killer whales days to years after the use of implanted tags indicates complete healing with no signs of infection and negligible scarring (see Figures 5-7) (Robin Baird, Cascadia Research Collective, pers. comm.). We expect that southern resident killer whales would heal similar to other populations and ecotypes of killer whales. Disinfectants can be used to minimize risk of infection (Aguilar and Nadal 1984), and tag tips used in the proposed studies would be cleaned with soap, bleach, boiling, acetone, and Clidex© or gas sterilization would be applied to reduce the risk of infection. While it is recognized that there is a risk of infection at the tag site, this would be minimized by the use of sterile procedures at all times and the smallest tag possible.

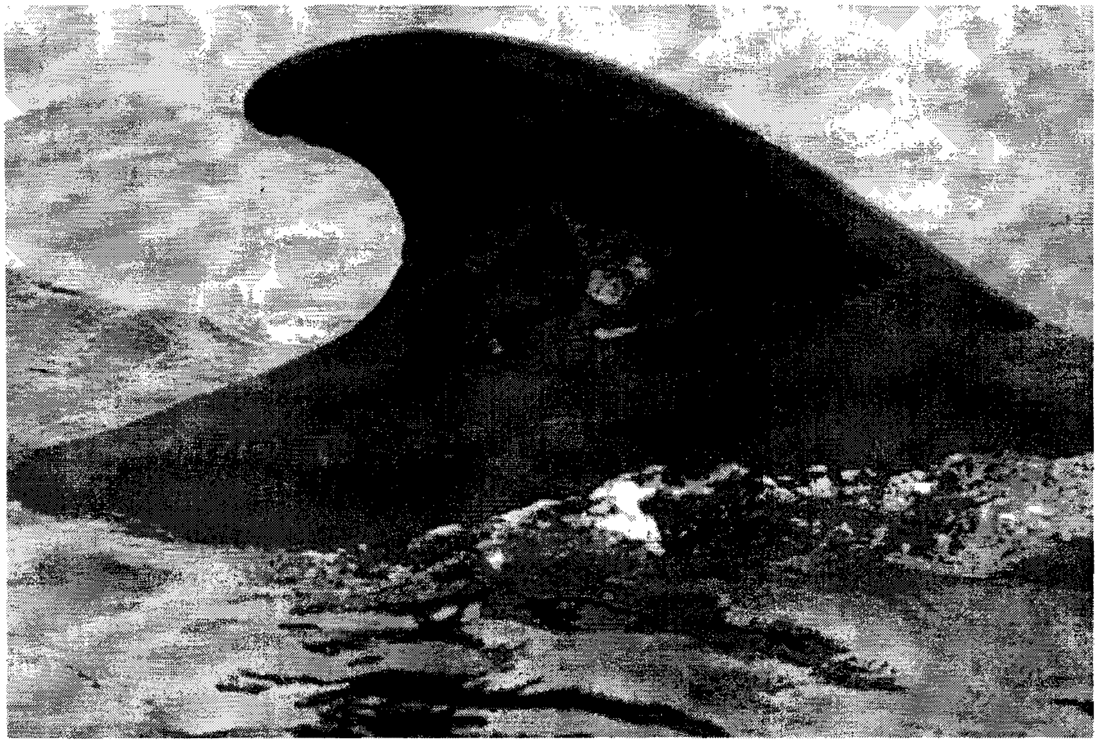

Figure 5. Tagging site on pilot whale tagged 41 days after tagging with similar implantable tag as that proposed by the applicants. Photo credit: Dan McSweeney under permit 782-1719.

Figure 6. Tagging site of same pilot whale 138 days post-tagging. Photo credit: Dan McSweeney under permit 782-1719.

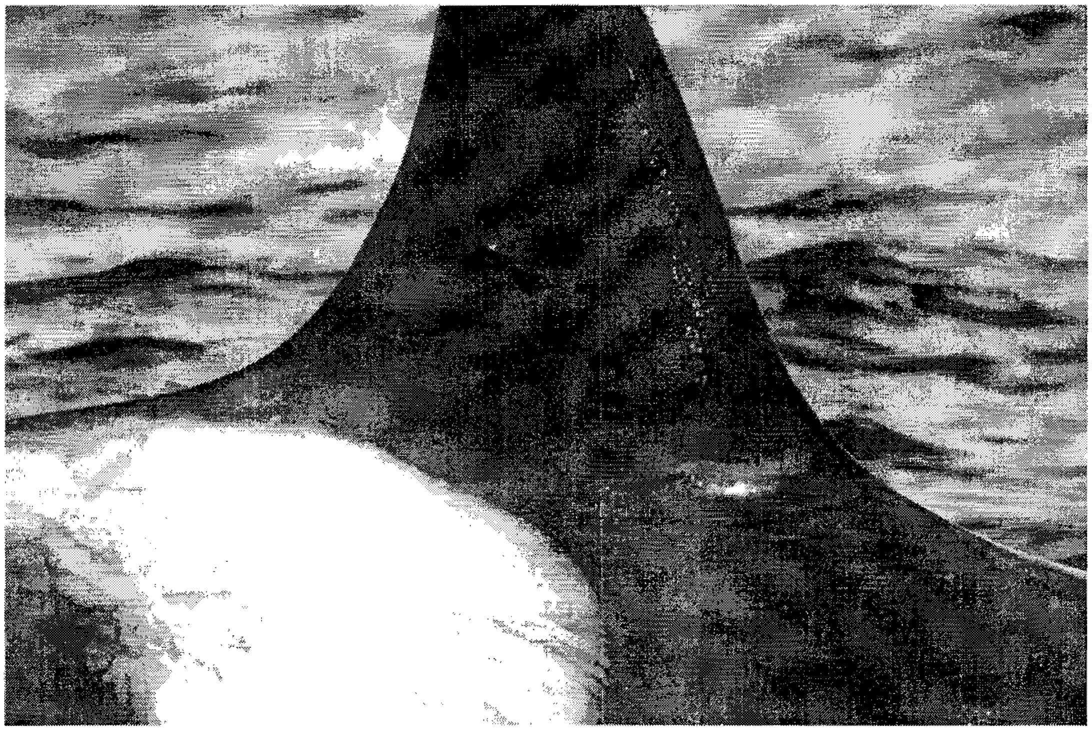

Figure 7. Tagging site on transient killer whale tagged 547 days after tagging with similar implantable tag as that proposed by Dr. Hanson. Photo credit: Maya Sears under permit 781-1824-02.

Risk of significant bleeding during the tagging of killer whales is thought to be minimal. Andrews et al. (2005b) concluded that the vasculature in the dorsal fin is likely controlled by sphincters and that the clotting and vessel healing properties would prevent any significant bleeding. Redundancy of vessels in the dorsal fin is expected to ensure thermoregulatory function even if a central vessel is compromised (Andrews et al. 2005b). In addition, the account in Erickson (1978b) indicated that even after four surgical pins pierced through the base of the dorsal fin, no bleeding occurred at the pin sites. Bleeding during invasive techniques has occurred, though. Andrews et al. (2005b) reported that killer whales in Norway and Iceland that had holes drilled into their dorsal fins sometimes experienced profuse bleeding; however, the bleeding ceased very quickly.

Whether any long-term effects resulting from tagging remain largely unknown and available information is limited. No research has been done to specifically assess the long-term impacts of tagging on killer whales. However, as described above, data from the resighting 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 2008b)).

To minimize the effects of dart tagging on southern resident killer whales, the permit requires researchers to only tag a killer whale in the dorsal fin. In addition, if a tag site becomes infected or does not heal properly, that animal may not be re-tagged at any time. A tagging attempt must be discontinued if an animal exhibits a strong adverse reaction to the activity or the vessel (such as breaching, tail lobbing, underwater exhalation, or disassociation from the group). Researchers would also apply "good practice" measures to minimize potential risks associated with tagging. Only qualified, experienced personnel with sufficient experience would perform the attachment of intrusive tags; that the attachment of invasive scientific instruments include the use of stoppers (in this case the tag itself acts as a stopper) to reduce the force of impact and limit the depth of penetration; and the use of the smallest possible tag size to minimize the potential for increased energetic costs of or behavioral responses to larger tags. Permit conditions to minimize the effects of tagging also address the potential for repeat disturbance of these species. The proposed permit limits tagging attempts on an individual to no more than twice daily and four times a year. The permit also requires coordination of the proposed activities with other permit holders conducting similar activities on the same species in the same locations or times of year.

Although not observed in southern resident killer whales, cetaceans of other social odontocete species have been known to remove tags from conspecifics or otherwise manipulate the tags (Irvine et al. 1979; Scott et al. 1990; White Jr. et al. 1981)(Robin Baird, Cascadia Research Collective, pers. comm.). This introduces the possibility of breakage of the tag and/or possible tag/component ingestion. We are not aware of a tag or its components being ingested, but we also lack information that documents this possibility. However, southern resident killer whales and virtually all other odontocete species are regularly exposed to marine debris of various types (both floating on the surface and items on the seafloor) that can be a proxy for the species propensity to ingest foreign objects. Ingestion of marine debris has not been documented in killer whales of the Pacific Northwest apart from ingestion of tags presumably attached to seals that were predated upon by transient killer whales (Brent Norberg, Lynne Barre, and Kristin Wilkinson, NMFS, pers. comm..s) and, based upon this, we expect the possibility of an individual removing and ingesting a telemetry tag to be very small.

We expect tag breakage may occur in some cases or situations. This is not unique to southern resident killer whales and was documented in other marine mammal taxa (Irvine et al. 1979; Scott et al. 1990). Although portions of the tags sometimes remained in the individual, where this has been investigated, the pieces were encapsulated and did not become infected (White Jr. et al. 1981). In another individual there was concern that a barb (unlike that proposed for use by the applicant) migrated further into the individual after its external components broke off, but no evidence to support or refute this was evident (White Jr. et al. 1981).

Two transient killer whales tagged with similar dart tags as those proposed for use under the proposed permit in 2010 have experienced significant wounds as a result of tagging and subsequent tag breakage. Follow-up photographic monitoring and review by expert marine mammal veterinarians, pathologists, and clinicians supports regional infection on the tag site over several months. The ultimate fate of broken tag components and, ultimately, the fate and fitness of the tagged individuals remains unclear at present, although poor overt health has not yet been observed and reproductive impacts are unknown. As a result of these instances, the manufacturer developed additional titanium plates and titanium nuts that tags proposed for use under permit 16163 would be outfitted with. Testing by Dr. Russ Andrews has shown significant reductions in breakage potential of barbs compared to previous models (Russ Andrews, Cascadia Research Collective, pers. comm.). As a consequence, we do not expect breakage of barbs to occur under permit 16163.

After reviewing available information on the responses of killer whales to implantable tagging procedures, we do not expect any mortality to occur due to the tagging under permit 16163. Injury from the implantable tags would be small and localized on the dorsal fin of targeted whales and is expected to heal completely and not result in any significant long-term physical damage. Rates of wound healing are expected to vary across regions and are not easily predicted in advance of the proposed tagging studies; however, photo and video monitoring would be conducted simultaneously and aid in determining wound healing or infection rates during the proposed studies. Resighting of tagged whales is expected to occur multiple times during the year and over the permit's life, facilitating monitoring of tag sites for years afterwards. Individuals in this DPS are typically monitored and photographed dozens of times each year, allowing substantial opportunity for post-tagging monitoring. We do expect up to four individuals would likely respond to tagging attempts annually with ephemeral startles and/or and dives based upon response rates observed by Andrews et al. (2005b).

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 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 occur remains 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. 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. Overall, monitoring report data from the applicant indicate fast dive, flinch, and tail slaps as responses of killer whales.

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 2008b)).

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. (2010a) 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 responded to hits with low to moderate-level reactions, but returned to baseline behavior within minutes (Hooker et al. 2001a). 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. 1998). 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.

Review of the literature and discussion with experts supports responses and response rates by target species to be generally similar between dart and suction cup tagging. Response data provided by Mate et al. (2007a) for blue, fin, right, and sperm whales appears to be the best source to appraise the rate of response by these species to suction cup 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. Based upon these response rates and the expected level of tagging, we expect that blue (16111- up to eight responses annually, 16163- up to one response annually), fin (16111- up to 23 responses annually, 16163- up to two responses annually), sei (16111- up to two responses annually), sperm (16111- up to 26 responses annually, 16163- up to nine responses annually), southern resident killer whales (16163- up to five responses annually), and Hawaiian insular false killer whales (16163- up to 28 responses annually), to respond to suction cup tagging activities with low- to moderate-level behavioral responses described above. 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 Tables 29-32 on pages 125-127).

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.

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 biopsy. Gauthier and Sears (1999) summarized data for several species, including blue, fin, and humpback (Table 34). Blue whales response 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 34. 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 34). 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).

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).

As with tagging activities, applicant 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 (2010b) are used to calculate bowhead response rate and Brown et al. (1991a) was used for right whales. Barrett-Lennard et al. (1996) was used to calculate response rate for southern resident killer whales. Overall, we expect blue (permit 16111-41 responses, permit 16163- one response), fin (permit 16111- 41 responses, permit 16163- two responses), sei (permit 16111- one response), humpback (permit 16111- 28 responses, permit 16163- two responses), sperm (permit 16111-10 responses, permit 16163-30 responses), and southern resident killer whales (permit 16163-21 responses) are likely to respond behaviorally to biopsy activities as described above (mild- to moderate-behavioral responses). As previously mentioned, individuals re-exposed to proposed activities could also undergo additional responses. Tables 29-32 on pages 125-127 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. Based upon data from Gauthier and Sears (1999), a few humpback whales may show strong but ephemeral behavioral reactions.

Pile driving experimental playback

Listed fishes

Very little is known about the effects of pile driving on fish in general and salmonids and rockfishes specifically (Popper and Hastings 2009). Increased fish mortality has been found within 50 m of pile driving; hemorrhage and swim bladder damage was observed post-mortem (PIDP 2001). Unfortunately, sound exposure levels were not available from this study and the relevance to the proposed action cannot be clearly drawn. Another study found fish caged at closer distances to pile driving generally had more tissue damage than those further away, although there was significant variability between individuals in the same test cages and confounding factors impacting sound exposure levels were not well controlled. However, individuals receiving sound exposure levels of 183 dB re: 1 uPa were generally found to not have tissue damage and individuals exposed to levels above 193 dB re: 1 uPa generally did (Abbott and Bing-Sawyer 2002 in Popper and Hastings (2009)). Nedwell et al. (2003) in Popper and Hastings (2009) found that brown trout (taxonomically similar to salmonids) did not respond behaviorally when caged and exposed to impact or vibratory pile driving 400 m away. CALTRANS (2004) in Popper and Hastings (2009) caged rainbow trout between 34 and 314 m from a pile driving source, exposed animals to pile driving as well as control treatment, and sacrificed individuals for pathological assessment. Although methods to evaluate damage were questionable and statistical analysis was not conducted, fish closer to the source generally had more tissue damage than those further away or control animals. Nedwell et al. (2006) in Popper and Hastings (2009) exposed brown trout to impact and vibopiling at extrapolated received levels of 193-201 dB re: 1 uPa for 200 m and found no external injury or damage to the internal ear.

Several studies specific to salmonids have been published. Feist (1991) found the number of pink and chum salmon schools in a cove isolated from pile driving noise to the number of schools in a construction area was about 2:1 on pile driving days and 1:1 on non-pile driving days. Feist (1991) qualitatively observed that juvenile pink and chum salmon exposed to pile driving noise were less apt to startle when approached by observers compared to schools in an acoustically-isolated cove. In another study, fish survey transects using a fathometer indicated schools of fish did not move away from the pile being driven or the general area of the pile driving barges during pile driving operations (PIDP 2001). Abbott et al. (2005) in Popper and Hastings (2009) conducted a more rigorous study using Chinook salmon caged ~10 m from a concrete pile being driven by an impact hammer and exposed for 200 hammer strokes over four minutes. No differences in mortality or injury were found. Perhaps the best available information regarding the behavioral impact and injurious potential of pile driving to listed salmonids stems from a study by (Ruggerone et al. (2008) in Popper and Hastings 2009). Here, yearling Coho salmon were placed in cages 2-15 m from an impact pile drive set up and control individuals were placed in ambient noise conditions far from the experimental site. Sound pressure levels were measured over exposures of 1,627 strikes over 4.3 hours, leading to received peak sound levels of up to 208 dB re: 1 uPa, sound exposure levels (SEL) of 179 dB re: 1 μPa2-s, and cumulative SELs of up to 207 dB re: 1 μPa2-s over the duration of exposure in the cages closest to the pile drive. No mortality or gross pathology were observed. Closed-circuit television was also used to observe behavior during the experiment and no significant changes in behavior were observed.

A multi-agency working group of Federal and State transportation and resource agencies, including underwater acoustics experts, fish biologists, and transportation specialists, has released agreed-upon "interim criteria" for evaluating the potential for physical effects (i.e., injury) from underwater noise levels caused by pile driving. These criteria represent threshold values for received levels, with the onset of injury expected if either: 1) the peak pressure of any strike exceeds 206 dB (re: 1μPa); or 2) SEL, accumulated over all pile strikes, exceeds 187 dB (re: 1 μPa^{2}·sec) for fishes 2 g or larger and 183 dB for fishes smaller than 2 g (FHWG 2008).

Based upon this evidence, we do not expect serious injury or mortality to any listed teleost under the NMFS' jurisdiction. It is reasonably likely that some avoidance of the source will occur and that stress in exposed individuals could increase. However, the longest the source would be active would be three minutes for acoustic playback, and this would involve a ramp-up procedure that would only allow fishes to be exposed to intensities that could produce behavioral responses for a period of less than one minute at a time. If behavioral responses were to occur, they would be temporary. If displacement or foraging were also affected, which are also reasonably possible, we expect these responses would also be temporary and that individuals would resume baseline activities shortly after exposure ends. Stress responses might be detectible for periods of hours to days, but we do not expect any stress response to rise to a level that is significant to the survival, growth, or reproductive potential of any individual.

Listed marine mammals

Studies specific to the response of killer whales or mysticetes to pile driving are not available (and, indeed, are the objective of the proposed active playback research), but studies of porpoises have been undertaken. Tougaard et al. (2009) found that harbor porpoises at least 21 km from a pile driving source altered their vocal behavior, presumably in response to impact pile driving.

The NMFS is currently developing comprehensive guidance on sound levels likely to cause injury and behavioral disruption in the context of the Marine Mammal Protection Act. Until formal guidance is available, NMFS uses conservative thresholds of sound pressure levels from broad band sounds that cause behavioral disturbance (160 dB rms re: 1 μPa for impulse sound and 120 dB rms re: 1 μPa for continuous sound) and injury (180 dB rms re: 1 μPa for whales and 190 dB rms re: 1 μPa for pinnipeds) (70 FR 1871).

Based on these conservative thresholds, the proposed acoustic playback activities could not produce sound pressure levels capable of injuring marine mammals. Although there are no studies that have analyzed behavioral responses of killer whales to pile driving, exposure of southern resident killer whales to sound at or above 120 dB may elicit behavioral responses within the range of previously documented responses by mid frequency hearing specialists to non-pulse sound. Southall et al. (2007) conducted a comprehensive literature review of the effects of sound on marine mammals. Behavioral responses in mid-frequency cetaceans from exposure to non-pulse sound can include moderate changes in speed of travel, direction, or dive profile; moderate to extended cessation or modification of vocal behavior; minor or moderate avoidance of the sound source, and change in group distribution (Southall et al. 2007). However, the authors caution there is considerable variability in received levels associated with behavioral responses that do not lead to clear conclusions. Contextual variables (i.e., novelty of the sound and what the animals are doing in the area) are the likely reason for the variable responses observed in mid-frequency cetaceans exposed to non-pulse sounds (Southall et al. 2007).

Due to the lack of prior data on killer whales or an appropriate surrogate, we are unable to specify a response or responses that we expect will occur based upon available information. We can, however, assess the likely response of exposed individuals based upon two lines of evidence: 1). A generalized response that mammals exhibit to novel anthropogenic stimuli and 2). A universe of possible responses and how significant these responses may be based upon the species' biology and limitations of permit 16163. At the beginning of this *Response analysis*, we presented information regarding mammalian responses to anthropogenic stressors; in general, observed responses are similar to those observed to predators (Beale and Monaghan 2004; Frid 2003; Frid and Dill 2002; Gill et al. 2001; Harrington and Veitch 1992; Lima 1998; Romero 2004). This often involves greater or lesser degrees of a stress response that can interrupt essential behavioral or physiological events, alter an animal's time budget, or constitute some combinations of these responses (Frid and Dill 2002; Romero 2004; Sapolsky et al. 2000; Walker et al. 2005). However, southern resident killer whales are not exposed to predation, as they are among the top-level marine predators in their ecosystem. Therefore, we do not expect distress as a result of exposure to limited-duration playbacks at the proposed sound intensities, as the proposed acoustic playback would not represent a predator and we have eliminated possibility of direct injury due to sound level exposure. Even if these were to occur, the proposed permit is conditioned to cease playbacks if "an animal exhibits repetitive strong adverse reactions to the playback activity or the vessel (e.g. aggression, aversion, avoidance, or extensive/prolonged changes in group cohesion)." Additionally, permit 16163 is conditioned that an individual can only be exposed to playback experiments once per day, meaning that the group associated with the exposure could also not be exposed again the same day. However, this does not fully address the possibility that the acoustic playback can be perceived as a threat, such as a stimuli that limits communication, leads prey to be more attentive or temporarily leave the area, or the significance to exposed individuals. We expect that all of these perceptions would lead to a slight to mild stress response and may also lead individuals to respond behaviorally. If perceived as a threat, we expect behavioral responses may include temporary displacement, signs of aggression or threat display, alteration of vocal and/or surface-active behavior, cessation of foraging and social activity (possibly including mating and group splitting), and/or protection of young individuals. However, permit 16163 requires that vessel approaches (and the activities associated with it) be discontinued if the approach appears to interfere with "vital functions," including reproduction and feeding. Based upon the aforementioned permit conditions, most of these behavioral responses would result in the shutdown of playback activities. The potential for vocal alteration and masking of communication, a non-rapid or prolonged movement away from the source vessel, and altered time and energy use budget still remain. We expect all of these, if they occur, to be temporary in time and space, as the playback would last a maximum of three minutes, the source level would rapidly attenuate with distance from the source, and we do not expect the source to be perceived as a major threat that would necessitate a prolonged response by exposed individuals.

We also considered the potential for other responses. Exposed individuals may perceive the playback signals as a neutral factor in the environment that does not justify a response or distract individuals from their baseline state. In this case, we do not expect an adverse affect to southern resident killer whales. It is also possible that playbacks may be perceived as a novel stimulus that elicits a response involving exploration of the source or the source vessel. In this case, we would not expect a stress response, but some alteration of an individual(s) time and energy budget as well as changes in vocal and/or social activity may be involved with gathering information on the stimuli. Approach of the source hydrophone and source vessel is possible. Because the permit is conditioned such that approaches cannot alter foraging or reproduction, nursing could not be interrupted because nursing calves could not be exposed, and exposed individuals would be engaged in a voluntary decision as to whether baseline activity or investigation of the playback is more preferable, we do not expect that this type of response would adversely affect any individual's ability to survive, grow, or reproduce.

We cannot identify the number of responses that individuals will likely have to the proposed playback activity. Until we receive information from subsequent monitoring and peer-reviewed literature based upon the proposed study, we provisionally assume that all individuals will respond in the most significant manner likely (temporary displacement, change in vocal behavior, and masking of relevant acoustic information).

The proposed acoustic playback activity would occur within the critical habitat of southern resident killer whales. The only primary constituent element that could be affected by this activity is that which protects the quality and quantity of southern resident killer whale prey. As we discussed in the previous section, salmonids may be displayed temporarily from the area immediately surrounding the acoustic playback source. However, this displacement is expected to be ephemeral and not significantly alter the quantity or quality of prey available to southern resident killer whales. We there for find that destruction or adverse modification of southern resident killer whale critical habitat is insignificant.

Prey mapping

Marine mammal response to prey-mapping sonar. We expect listed marine mammals to experience ensonification from systems used to image prey. These systems resemble multibeam echosounders and are much higher than frequencies vocalized or heard by all listed whales except humpback, sperm, and southern resident killer whales (Steller sea lions may be capable of hearing signals as well). Although Todd et al. (1992) found that mysticetes reacted to sonar sounds at 3.5 kHz within the 80-90 dB re 1 μPa range, it is difficult to determine the significance of this because the source was a signal designed to be alarming and the sound level was well below typical ambient noise.

Hearing is poorly understood for listed baleen whales, but it is assumed that they are most sensitive to frequencies over which they vocalize, which are much lower than frequencies emitted by the proposed multibeam echosounder (Ketten 1997; Richardson et al. 1995b). Thus, if blue, fin, sei, or North Pacific right whales are exposed, they are unlikely to hear these frequencies well (if at all) and a response is not expected.

Assumptions for humpback and sperm whale hearing are much different than for other listed whales. Humpback and sperm whales vocalize between 3.5-12.6 kHz and an audiogram of a juvenile sperm whale provides direct support for hearing over this entire range (Au 2000; Au et al. 2006; Carder and Ridgway 1990; Erbe 2002a; Frazer and Mercado 2000; Goold and Jones 1995; Levenson 1974; Payne and Payne 1985; Payne 1970; Richardson et al. 1995b; Silber 1986; Thompson et al. 1986; Tyack 1983; Tyack and Whitehead 1983; Weilgart and Whitehead 1993; Weilgart and Whitehead 1997; Weir et al. 2007; Winn et al. 1970). Auditory brainstem response and behavioral response studies of captive killer whales support their ability to hear signals in the frequency range proposed to be used for prey mapping (Szymanski et al. 1995a; Szymanski et al. 1999; Szymanski et al. 1998; Szymanski et al. 1995b). Maybaum (1990; 1993) observed that Hawaiian humpbacks moved away and/or increased swimming speed upon exposure to 3.1-3.6 kHz sonar. Kremser et al. (2005) concluded the probability of a cetacean swimming through the area of exposure when such sources emit a pulse is small, as the animal would have to pass at close range and be swimming at speeds similar to the vessel. Sperm whales have stopped vocalizing in response to 6-13 kHz pingers, but did not respond to 12 kHz echo-sounders (Backus and Schevill 1966; Watkins 1977; Watkins and Schevill 1975). No information is available regarding killer whale responses to echosounders, but we expect that the possible responses exposed individual may exhibit would be consistent with what has been observed in humpback and sperm whales (vocal behavior changes and temporary displacement). Furthermore, proposed permit 16163 is conditioned such that if disturbance in vital activities such as feeding or breeding is observed, researcher activities would be discontinued.

We do not expect masking of humpback, sperm, or southern resident killer whale communications to appreciably occur due to multibeam echosounder signal directionality, low duty cycle, and the brief period when an individual could be within its beam.

Based upon this, we expect that humpback, sperm, and southern resident killer whales may alter vocal behaviors or be temporarily displaced from a location in a subset of cases where they are exposed to multibeam echosounder signals. As the source level is relatively low, these signals should attenuate rapidly and not be audible to listed cetaceans over a wide distance.

No information is available regarding the effect that echosounder systems might have on Steller sea lions. However, Steller sea lions are not known to echolocate, home in on prey using sound, or communicate vocally underwater and thus these processes would not be disturbed as they might be in cetaceans. If a Steller sea lion were to approach the echosounder closely, it is expected that the individual would move a short distance away as a result if any discomfort were experienced. Repeated exposure is not expected and subsequent response is not expected. We expect exposures and subsequent responses of this kind to be very rare.

Recent stranding events associated with the operation of naval sonar suggest that mid-frequency sonar sounds may have the capacity to cause serious impacts to marine mammals. The sonars proposed for use by the applicants differ from sonars used during naval operations, which generally have a longer pulse duration and more horizontal orientation than the more downward-directed multibeam echosounders. The sound energy received by any individuals exposed to the multibeam echosounder source during the proposed activities is lower relative to naval sonars, as is the duration of exposure. The area of possible influence for the multibeam echosounder is also much smaller, consisting of a narrow zone close to the source vessel. Although navigational sonars are operated routinely by thousands of vessels around the world, stranding incidence of has been correlated to use of these sonars. Because of these differences, we do not expect these systems to contribute to a stranding event.

Ultrasound

Permit 16163 would authorize the use of ultrasound to assess blubber thickness of free-ranging southern resident killer whales. We cannot identify any information to help establish what response(s) have been in this species while in the wild and instead draw from similar research on other free-ranging marine mammals to establish probable response(s). The first use of ultrasound on free-swimming cetaceans was conducted in 1998 on North Atlantic right whales for similar reasons as the proposed research (assess body condition) and later in southern right whales (Angell et al. 2004; Moore 1998; Moore et al. 2001; Watts and Harvey 2005). Although these reports do not address the responses that individuals exhibited to an ultrasound sensor being placed along their dorsum as they surfaced, a principal investigator indicated that targeted individuals were indifferent, moved slowly away from the vessel once the ultrasound probe contacted the whale, or flexed their backs (Michael Moore, WHOI, pers. comm.).

The methods proposed to be used in using ultrasound on southern resident killer whales are similar, but not as invasive as those used in dart and suction cup tagging or biopsy attempts. We do not expect responses to be more significant than these and probably less significant in terms of response intensity, although we have no estimate of what response frequency might be. Based upon this information, we expect responses to ultrasound attempts to be similar to those observed in other activities involving brief physical contact with southern resident killer whales in conjunction with very close vessel approach (flinch, shake, fast dive, tail slap, roll, and/or acceleration). We provisionally assume that all individuals will respond in these ways until we obtain better information documenting response and response rate from monitoring reports and peer-reviewed publications produced as a result of the proposed ultrasound activities.

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 or proposed 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.

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. 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). 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. 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/). Population growth may increase toxic runoff and hard surface area that facilitate the runoff, reduced oxygen levels 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 2008f; Ross et al. 2000b). 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 2008f).

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 or proposed 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 or proposed 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 permits to Dr. Kenneth Balcomb III (15569), Dr. John Calambokidis (16111), Dr. Jenny Atkinson (16160), and Dr. Brad Hanson (16163) for directed take of listed and proposed for listing cetaceans in the Pacific Ocean (blue, fin, sei, humpback, North Pacific right, sperm, southern resident killer whales, and Hawaiian insular false killer whales), 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), Hawaiian monk seals (endangered), as well as Guadalupe fur seals (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. 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. 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 Pacific Ocean are expected face area-specific threats identified in the *Environmental baseline*, including habitat degradation, whale watching, research activities, naval activities, climate change, human noise sources, ship strike, 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 southern resident killer whales have recently fluctuated and are mostly in decline. Insular Hawaiian false killer whales appear to be either stable or in decline. 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 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 25-28 on pages 120-124 for expected exposure).

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. 1984; Richardson et al. 1985b; Scheidat et al. 2004)). 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). The presence of additional anthropogenic stressors, such as commercial vessel traffic, are likely to induce additional disturbance on potential target individuals (Fraker et al. 1982; NMFS 2008a). For all but southern resident killer whales, the number and severity of responses to research vessel approaches that listed individuals will experience is negligible. This is especially so in comparison to the other anthropogenic and natural stressors that individuals must cope with. Although it is possible that individuals are being displaced from more preferable habitat, we have no evidence to suggest this. On the contrary, Weinrich (2010) found that individuals exposed to extensive whale watching retained high calving rates and positive population trajectories. 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.

We expect southern resident killer whales to receive far more frequent exposure to approach per individual than any other listed species under the proposed permits. Our *Exposure analysis* supports that a given individual is reasonably likely to be approached by research vessels of the proposed permits up to 92 times annually, although it is most likely that this number would be 65 times annually. Information from our response analysis indicates that southern resident killer whales rarely, if ever, respond subtly or overtly to individual research vessel approaches when no other activity, such as tagging or biopsy, occurs concurrently. However, there is also significant information to support that responses do occur, including altering vocal activity, altered direction of travel, mother-calf separation, and possibly influencing foraging success (Bain et al. 2006b; Erbe 2002d; Foote et al. 2004b; Williams et al. 2002b; Williams et al. 2006; Williams et al. 2002d); these reactions have been observed exclusively or nearly so in association with recreational or commercial whale-watching vessels (NMFS 2006b). The vast majority of vessel operating near southern resident killer whales are commercial and recreational vessel traffic that target members of this species for whale watching, the exposure to which we cannot quantitatively separate from research-based work such as that proposed to be permitted (Hoyt 2001; Hoyt 2002; Koski 2006a; Koski 2007b; MMMP 2002b; NMFS 2008f; Osborne 1991). However, we do expect that responses are due to the cumulative exposure to vessel traffic that southern resident killer whales experience and/or provocative approaches of commercial and/or recreational vessels and have little contribution in frequency or intensity from research-based vessel approaches (Koski 2005; Koski 2008; Koski 2010; Koski 2011). We expect observed responses to become less frequent and significant in the future due to new mandatory guidelines the NMFS has established for vessels operating near southern resident killer whales (76 CFR 20870). This cumulative information leads us to conclude that responses by southern resident killer whales will not be any more significant than the few potential low or moderate level responses we expected from the *Response analysis*.

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 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. 2001b; Mate et al. 1997b; Watkins 1981c; Watkins et al. 1984b). In addition, not all individuals respond to tagging, meaning that a fraction of targeted individuals are not expected to show an overt response to the combined approach and tagging action. We do expect all individuals to at least be aware of the vessel's approach and undergo a low-level 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) 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. 2007b; Weller 2008). At present, available evidence from a single animal of advanced decomposition, a single observation of a deceased North Atlantic right whale and numerous observations of live whales does not support debilitating infection caused by implantable tags (McLellan 2011; Weller 2008). Until additional tagging sites of have been evaluated, the issue will likely remain unresolved. Methods adopted by the applicants to be authorized for tagging, including use of disinfectants on tagging materials penetrating target individuals, should minimize the risk of infection (Mate et al. 2007b; 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. 2007b; 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. 2007b), we do not expect these physiological responses to be significant to any individuals' overall metabolic balance or health state.

Simulated pile driving may cause southern resident killer whales to experience minor stress responses and undergo temporary, minor behavioral responses as a result of acoustic playbacks; very few Steller sea lions may unintentionally experience the same effects. Some listed fishes may also respond in similar ways. Echosounding for prey mapping may cause similar effects in southern resident killer whales and humpback whales. Ultrasound placement on southern resident killer whales is also expected to cause ephemeral behavioral and stress responses in a subset of exposed individuals.

Overall, we expect all targeted whales to experience some degree of stress response to approach 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. 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 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, North Pacific right, sperm, southern resident killer, and Hawaiian insular false killer whales (proposed as endangered), threatened eastern DPS Steller sea lions, threatened Puget Sound and

Hood Canal summer-run chum salmon, threatened Puget Sound steelhead, endangered bocaccio, threatened yelloweye rockfish, and threatened canary rockfish 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 permits 15569, 16111, 16160, and 16163 are likely to adversely affect individuals of the aforementioned 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 recommendation 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.* The Endangered Species Act Interagency Cooperation Division recommends that the Permits Division should examine its methodologies for determining take numbers and coordinate with the Endangered Species Act Interagency Cooperation Division to ensure that the take numbers better reflect a level of exposure which has occurred in the past under similar or identical researcher actions as evidenced by annual reports.
2. *Identify responses by listed individuals to permitted actions.* The Endangered Species Act Interagency Cooperation Division recommends that annual reports submitted to the Permits Division require detail on the response of listed individuals to permitted activities. A minimum of general comments on response can be informative regarding methodological, population, researcher-based responses in future consultations. The number and types of

responses observed should be summarized and include responses of both target and non-target individuals. This will greatly aid in analyses of likely impacts of future activities.

In order for NMFS Endangered Species Act Interagency Cooperation 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 Act Interagency Cooperation Division of any conservation recommendations they implement in their final action.

Reinitiation Notice

This concludes formal consultation on NMFS' proposal to issue permits 15569, 16111, 16160, and 16163, 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 Division must immediately request reinitiation of section 7 consultation.

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