National Marine Fisheries Service
Endangered Species Act Section 7 Consultation

Biological Opinion

Agency: United States Department of Homeland Security, United States Coast Guard

Activities Considered: An assessment of the United States Coast Guard's National Ballast Water Management Program and Initial Numerical Standard

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

Approved by:

Date: JUN 20 2012

Section 7(a)(2) of the Endangered Species Act (ESA), as amended requires each Federal agency to insure that any action they authorize, fund, or carry out 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 (16 U.S.C. 1531 *et seq.*). When the action of a Federal agency "may affect" a threatened or endangered species or critical habitat that has been designated for them, that agency is required to consult with either the National Marine Fisheries Service (NMFS) or the U.S. Fish and Wildlife Service (USFWS) (together, "the Services"), depending upon the species that may be affected by the action. In this case, NOAA Fisheries conducted formal programmatic consultation with the U.S. Coast Guard on their ballast water management program to determine whether the program complies with the requirements of section 7(a)(2) of the Endangered Species Act (that is, the requirement to insure that their actions are not likely to jeopardize the continued existence of endangered or threatened species or result in the destruction or adverse modification of critical habitat that has been designated for those species).

This document represents NMFS' biological opinion (Opinion) on the United States Coast Guard's (USCG) ballast water management program, authorized under the Nonindigenous Aquatic Nuisance Prevention and Control Act of 1990 (NANPCA) as reauthorized and amended in the National Invasive Species Act of 1996 (NISA). Within this program, the USCG is proposing to establish, monitor, enforce, and if practicable and determined to be necessary, revise, a ballast water standard. Establishment of this standard is intended to ensure to the maximum extent practicable that aquatic nuisance species are not discharged into waters of the United States from vessels. The purpose of this consultation is to evaluate the ballast water management program and to analyze risks associated with introducing non-native species from discharged ballast water meeting the proposed numerical standard. This programmatic consultation assesses the probable exposure and possible risks associated with the ballast water management standard and then reach conclusions on the USCG's ballast water management program generally defined in the final rule (FR) for the allowable concentration of living organisms in ships' ballast water discharged in waters of the United States. While this action addresses ballast water management, non-native species can also be introduced by water held in the sea chest or by reproduction of organisms attached to the hull of the vessel that are not within the scope of this consultation.

This Opinion has been prepared in accordance with section 7(a)(2) of the ESA of 1973, as amended (16 U.S.C. 1531 et seq.) and implementing regulations at 50 CFR 402. However, consistent with a decision rendered by the Ninth Circuit Court of Appeals on August 6, 2004, we did not apply the regulatory definition of "destruction or adverse modification of critical habitat" at 50 CFR 402.02. Instead, we relied on the statutory provisions of the ESA to complete our analysis of the effects of the action on designated critical habitat. Essential fish habitat (EFH) consultations, in accordance with Section 305(b)(2) of the Magnuson-Stevens Fishery Conservation and Management Act (MSA) (16 U.S.C. 1801, *et seq.*) and implementing regulations at 50 CFR 600, are conducted at a regional level and are not within the scope of this consultation.

This Opinion is based on our review of the October 21, 2011, letter requesting initiation of consultation, the final Programmatic Environmental Impact Statement (PEIS), draft PEISs, the FR, and supporting documentation; the draft U.S. recovery plans for Sacramento winter-run Chinook salmon, Central Valley spring-run Chinook salmon, Southern California steelhead, and California Central Valley steelhead trout; the final U.S. recovery plans for smalltooth sawfish, Atlantic salmon, Upper Columbia River spring Chinook salmon, Puget Sound Chinook salmon, Hood Canal chum salmon, gulf sturgeon, shortnose sturgeon, Ozette Lake sockeye salmon, Upper Columbia River steelhead, Middle Columbia River steelhead, blue whale, fin whale, humpback whale, southern resident killer whale, North Atlantic right whale, sperm whale, Hawaiian monk seal, eastern and western Steller sea lions, Northwest Atlantic population of loggerhead turtles, Pacific populations of the loggerhead turtle, Atlantic populations of green turtles, Pacific populations of the green turtle, hawksbill turtle in the Atlantic, Pacific populations of the hawksbill turtle, leatherback turtle in the Atlantic, Pacific populations of the leatherback turtle, Pacific populations of the olive ridley turtle; the bi-national recovery plan for Kemp's ridley turtle; white papers; primary literature; past and current research, both published and unpublished; the documents that were used to list green sturgeon and smalltooth sawfish as threatened and endangered species (respectively); information pertinent to the performance of other USCG environmental enforcement programs; and monitoring reports from prior ballast water reporting and recordkeeping requirements.

Consultation History

Prior to discussing the final PEIS and potential impacts to NMFS trust resources, NMFS met with the USCG multiple times between 2006 and 2011. These meetings and comments were part of the pre-consultation process. While formal programmatic consultation was not initiated until November 22, 2011, the agencies shared draft letters requesting initiation that were important to the programmatic consultation process.

On January 31, 2011, NMFS received a draft letter from the USCG requesting initiation of consultation.

On March 9, 2011, NMFS provided a response to the USCG letter, providing a complete species list and requesting a consistent use of the environmental baseline for ESA purposes.

On March 29, 2011, NMFS and USCG had a teleconference to discuss ESA baseline and ballast water management standard impacts. NMFS stated that even if ballast water standards are an improvement over previous treatments, the draft PEIS did not support a conclusion of not likely to adversely affect listed species or their critical habitat. Previous determinations regarding ballast water exchange (BWE) had addressed uptake and physical impacts (thermal, salinity), but not effects from non-native species. During the meeting, we also discussed whether to address the proposed International Maritime Organization (IMO) standard or program including all feasible alternatives and the USCG's long term goal. NMFS recommended a programmatic approach because of the nature of the action.

On March 30, 2011, NMFS and USCG met to discuss edits made to the final draft PEIS in response to NMFS' comments and to establish the appropriate steps forward to initiate consultation.

On May 12, 2011, the USCG provided NMFS with a draft request for initiation and asked NMFS to provide comments.

On May 23, 2011, NMFS responded that the information in the letter did not fully reflect the outcome of the March 30 meeting and the effects analysis did not seem to support the conclusion reached in the letter. Therefore, on May 24, 2011, NMFS and the USCG met to discuss the initiation request letter.

On June 22, 2011, USCG provided NMFS with another draft reflecting agreements reached on May 24, 2011.

On August 1, 2011, NMFS provided additional comments to the USCG reflecting agreements reached on May 24, 2012, and clarified on June 22, 2012.

On August 12, 2011, the USCG provided a new draft with changes made in response to NMFS' comments. NMFS agreed with the draft.

On October 28, 2011. NMFS received the final, signed request for initiation of programmatic consultation from the USCG, dated October 21, 2011. In the supporting documentation for that request, the USCG determined their action "may effect" listed species and their critical habitat, but left it up to NMFS to determine which species would be adversely affected and which were not likely to be adversely affected.

On November 9, 2011, NMFS hosted a meeting with USFWS and USCG to ensure all three agencies agreed on the proposed action and proposed determination of effects.

A series of emails between November 14, 2011, and November 22, 2011, discussed NMFS' intentions of initiating formal consultation to produce an Opinion rather than providing a letter of concurrence.

On November 22, 2011, the USCG and NMFS agreed to formal programmatic consultation on the USCG ballast water management program as was requested in the October 28, 2012, letter (dated October 21, 2012).

On December 6, 2011, NMFS and USFWS attended a meeting at the USCG, where the USCG had several questions about the environmental baseline and how that led to a likely to adversely affect determination. There was also discussion about the ESA timeline and when consultation could be completed. Additionally, all three agencies agreed to meet on a biweekly basis so the USCG could be updated on the progress of the Opinions.

On December 20, 2011, NMFS, USFWS, and the USCG met. The main focus of the conversation was the scope of the proposed action. The USCG suggested the consultation should not be programmatic but rather only address the interim final rule. The USCG requested case law on segmentation of the consultation process. Because no agreement was reached on the scope of the consultation, both agencies agreed to discuss this topic after the holidays on January 5, 2012.

On December 21, 2011, the USCG requested that NMFS continue consultation under the programmatic scope as was initially requested.

On December 23, 2011, NMFS' Office of General Council provided examples of general ESA case law relevant to the ESA baseline and the consultation process.

On January 5, 2012, NMFS, USFWS, and USCG met with the main topic again being the scope of the proposed action. NMFS provided a summary of the proposed action and the USCG agreed to provide any comments to both agencies by January 9, 2012.

On January 17, 2012, NMFS, USFWS, and USCG met to discuss the continued disagreement on the scope of the proposed action. NMFS agreed to provide the Proposed Action section from the Opinion so both agencies could identify where the disagreement originated and whether the agencies could reach agreement. NMFS also requested information related to monitoring and the USCG provided information related to the oil water separator program which would be similar to the compliance monitoring plan in the FR.

On January 31, 2012, NMFS, USFWS, and USCG met to discuss the proposed action and effects analysis. NMFS also requested additional information about the monitoring program and agreed to present those questions in an email following the meeting.

On February 1, 2012, NMFS received comments on the proposed action section. NMFS addressed the comments, accepting much of the language and explaining the reason for differences in other places. NMFS provided USCG with another draft of the proposed action on Febraury 2, 2012.

On February 9, 2012, NMFS and USCG agreed on the proposed action section after a final delivery of comments and explanations by the USCG.

On March 6, 2012, NMFS, USFWS, and USCG met to update each other on the progress of the Opinion and FR. NMFS and USFWS both agreed they would have a draft of the Opinion ready to share on March 30, 2012.

On March 30, 2012, NMFS provided a draft Opinion to the USCG.

On April 18, 2012, the USCG provided NMFS with comments on the draft Opinion.

On May 8, 2012, after incorporating USCG's comments, NMFS provided the USCG with a second final draft.

On May 15, 2012, the USCG provided a second round of comments focusing on the difference between the practicability review and periodic review.

On May 24, 2012, NMFS and the USCG agreed on the portion of the proposed action that identifies the differences between the practicability review and periodic review.

Legal and Policy Framework for Ballast Water Regulation by the USCG

Under the Nonindigenous Aquatic Nuisance Prevention and Control Act of 1990 (NANPCA) as reauthorized and amended in the National Invasive Species Act of 1996 (NISA), the USCG is the lead federal agency in implementing regulations to reduce or prevent the introduction of nonindigenous species via shipping activities in United States (US) waters. The purposes of NANPCA are to (1) prevent unintentional introduction and dispersal of non-indigenous species (NIS) into waters of the US through ballast water management and other requirements, (2) coordinate federally conducted, funded, or authorized research; and to federally coordinate prevention control, information dissemination, and other activities regarding the zebra mussel and other aquatic NIS, (3) develop and carry out environmentally sound control methods to prevent, monitor, and control unintentional introductions of NIS from pathways other than ballast water exchange, (4) understand and minimize ecological and economic impacts of NIS that become established, including the zebra mussel, and (5) establish a program of research and technology development and assistance to states in the management and removal of zebra mussels. Under NANPCA, Congress directed the USCG to develop a national program of specific regulations and guidelines intended to prevent or reduce the introduction and control the spread of NIS by ensuring to the maximum extent practicable that NIS are not discharged into waters of the US from vessels equipped with ballast water tanks.

On April 8, 1993, the USCG published a final rule, "Ballast Water Management for Vessels Entering the Great Lakes" [58 FR 18330]. The final rule mandated the use of ballast water management practices by ships entering the Great Lakes after operating outside the exclusive economic zone (EEZ). On December 30, 1994, the USCG published a final rule, "Ballast Water Management for Vessels Entering the Hudson River" [59 FR 67632], which expanded the mandatory requirements for the Great Lakes to portions of the Hudson River (north of the George Washington Bridge) that connect to the Great Lakes. The management practices mandated for use in the Great Lakes and portions of the Hudson River are found in 33 CFR Part 151, Subpart C.

NISA reauthorized and amended NANPCA and mandated the USCG to issue ballast water management guidelines and regulations applicable to all US waters. Under NISA, Congress authorized the USCG to establish a mandatory ballast water management program [69 FR 44952], if shipping industry compliance under the initially established national voluntary guidelines was insufficient. NISA also authorized the environmentally sound alternative methods, if the alternative methods are at least as effective as ballast water exchange. This includes any method, effort, or program to prevent or reduce invasive species introductions or control infestations, that minimizes adverse impacts to the structure and function of an ecosystem, and minimizes adverse effects on nontarget organisms and ecosystems (USCG 2003).

On May 17, 1999, the USCG published the interim rule for the Implementation of the NISA [64 FR 26672] and finalized it on November 21, 2001 [66 FR 58381]. The rule set voluntary ballast water management guidelines applicable to all US waters, except for the Great Lakes and the Hudson River, where the mandatory requirements remained in effect, and mandated reporting requirements. Both the voluntary management guidelines and the mandatory reporting requirements applied to vessels entering US waters after operating in waters outside the EEZ. The rule also promoted ballast water management for all vessels operating within US waters (defined as the waters of all 50 states, the District of Columbia, Puerto Rico, Guam, American Samoa, the US Virgin Islands, and the Trust Territories of the Pacific Islands).

As a result of several years of public comments and intra-governmental meetings, the USCG published a notice on September 26, 2003 [68 FR 55559], announcing USCG’s intent to prepare a PEIS to assess the impacts of developing a ballast water management standard. In the meantime, the USCG also published the final rule on the mandatory ballast water management program for all US waters, under NISA, on July 28, 2004 [69 FR 44952]. This rule requires vessels that operate outside the US EEZ to use one of the following management practices: (1) conduct mid-ocean ballast water exchange 200 nautical miles from any shore, (2) retain ballast water onboard, or (3) use a USCG-approved alternative method.

Description of the Ballast Water Management Program

Since 2003, the USCG has been developing a ballast water management program in cooperation with other federal agencies. The program is described by the USCG in the October 21, 2011, letter requesting initiation of consultation, the FR, and the final PEIS and framed by the NANPCA and the NISA. The goal of the USCG’s ballast water management program is identified in NISA, and specified in the final PEIS, as preventing the unintentional introduction and dispersal of NIS into waters of the US through ballast water treatment and other requirements. This action will change language in 33 CFR 151 to reflect changes to the program. The USCG is also amending its regulations to establish a type-approval requirement for ballast water management systems, which will undergo, if necessary, separate consultation tiered from this programmatic consultation.

Actions Authorized by the Program

The USCG's ballast water management program proposes to establish a numerical standard and allows for a stricter standard to be established via a series of practicability reviews. The USCG's FR establishes the following ballast water management standard:

> (1) For organisms greater than or equal to 50 micrometers in minimum dimension: discharge must include fewer than 10 living organisms per cubic meter (m^3) of ballast water.
>
> (2) For organisms less than 50 micrometers and greater than or equal to 10 micrometers: discharge must include fewer than 10 living organisms per milliliter (mL) of ballast water.
>
> (3) Indicator microorganisms must not exceed:
>
> (i) For Toxicogenic *Vibrio cholerae* (serotypes O1 and O139): a concentration of less than 1 colony forming unit (cfu) per 100 mL.
>
> (ii) For *Escherichia coli*: a concentration of fewer than 250 cfu per 100 mL.
>
> (iii) For intestinal enterococci: a concentration of fewer than 100 cfu per 100 mL.

Practicability Review and Periodic Review

NISA requires the USCG to assess and, if necessary, revise USCG'sballast water management regulations not less than every three years based onthe best scientific information available at the time of that review (16 U.S.C.4711(e)(1). The FR establishes apracticability review, completed no later than January 1, 2016, to determinewhether technology and testing protocols are available to practicably implementa more stringent ballast water discharge standard. The practicability review (33 CFR 151.2030(c)) will evaluate the capability of any identified technology to achieve a more stringent ballast water management standard; the effectiveness of any identified technology in the shipboard environment; the compatibility of any identified technology with vessel design and operation; the safety of any identified technology; whether the use of any identified technology would adversely affect the environment; the cost and cost-effectiveness of any identified technology; the economic impact of any identified technology; the availability, accuracy, precision and cost effectiveness of methods to measure concentrations of organisms or chemicals per volume of treated ballast water; and any other factors the USCG considers relevant to the review.

Additionally, under NANPCA/NISA (16 U.S.C. 4711(e)(1)), the USCG will assess the effectiveness of the USCG's ballast water management regulations in reducing the introduction and spread of aquatic nuisance species by vessels and, as necessary and based on the best scientific information available at the time, revise the regulations and/or promulgate additional regulations pursuant to NANPCA/NISA. Understanding the best scientific information available changes with the advent of time, the periodic reviews will likely include assessments of the practicability of technology or ballast water testing capabilities that would allow for a more stringent ballast water discharge standard, including the capability of a new technology to meet a more stringent standard, the capability of testing methods to allow for enforcement of a more stringent standard, the effectiveness of approved technologies in the shipboard environment, and whether the use of any new technologies may have an adverse impact on the environment. To fulfill the NISA/NANPCA statutory mandate of ensuring to the maximum extent practicable that aquatic nuisance species are not discharged into U.S. waters from vessels equipped with ballast tanks, the periodic reviews will likely entail a broader perspective than the review under 33 CFR 151.2030(c), focusing not solely on the practicability of implementing a more stringent discharge standard, but including assessments of practicable ballast water treatment as well as other ballast water, vessel management, and compliance assessment practices that could be implemented to achieve greater reductions in the risks of vessel-mediated introductions of aquatic nuisance species.

Reporting and Record Keeping

Vessels continue to be required to report and maintain records of their ballast water management and be subject to onboard testing during port and flag state control examinations as well as annual inspections (16 U.S.C. 4711(c)(2)(F). These requirements are met by the reporting and recordkeeping regulations (33 CFR 151.2060 and 33 CFR 151.2070, respectively) and Shipboard Testing Requirements (46 CFR 162.060-28). Information on ballast water management must be retained by the owner and/or operator for 24 months, with the option of retaining information electronically for all but the most recent six months. The USCG has removed the ballast water reporting form from the CFR (previously an appendix to 33 CFR Subpart D) in order to streamline any future changes to the form. The form is still the proper form to satisfy the reporting requirements, but 33 CFR 151.2070 has been revised to reference the National Ballast Information Clearinghouse (NBIC) website as the form's location. The Coast Guard has initiated a separate rulemaking to revise the form as needed.

Each vessel will be required to record vessel information, voyage information, ballast water information, ballast water management information, information on ballast water to be discharged, information on sediment to be discharged, and certification that the information is accurate. The required vessel information will be the name of the vessel, that vessel's IMO number, vessel type, owner or operator, gross tonnage, call sign, and state of registry. The required voyage information will be date and port of arrival, the vessel agent, the last port of call, and next port of call. The required ballast water information will be total ballast water capacity, total volume of ballast water onboard, total number of ballast tanks, and total number of ballast tanks in ballast. The required ballast water management information will be the total number of ballast tanks/holds that will be discharged into waters of the US, all alternative ballast water management methods used for each ballast tank, and identify whether there is a ballast management plan onboard and whether it was used. The required information on ballast water to be discharged will be the origin of the ballast water, including dates, locations, volumes, and temperatures; treatment information will include dates, locations, volumes, method, percent exchanged (if using ballast water exchange), and sea height at time of exchange or treatment; and the expected date, location, volume and salinity of any water to be discharged. The required information on ballast sediment to be discharged will be the name and location of the facility where sediment disposal will take place.

Compliance Monitoring

The USCG identifies the components of their compliance monitoring in 33 CFR 151.2075. The compliance monitoring allows the USCG to verify that vessels are equipped with approved technology. This section of the FR also authorizes the USCG to board vessels and take samples of ballast water as well as ballast sediments to verify treatment technologies are being used to treat ballast water and to ensure vessels are meeting the ballast water numerical standard. This monitoring is established to function in essentially the same way the oil-water separator monitoring program works.

The USCG has been monitoring vessels under the ballast water management program since 1995. Compliance monitoring is an important component of that monitoring. To determine whether a vessel conducted BWE, which was the previous required ballast water treatment method, the USCG sampled ballast tanks to determine if the water in the ballast tank had been exchanged or if it was still from the original port. As is proposed in the FR, the USCG will monitor self-reports, records, and continue to conduct on-board monitoring to verify that ships are complying with the ballast water management program.

All vessels are required to install approved ballast water treatment technologies by the vessel's first scheduled drydock after January 1, 2016. If a ship is unable to install the technology by that date, they may seek an extension. The USCG will assess the reason behind the need for an extension and if warranted may grant up to an additional 12 months (dependent on vessel type and treatment technology to be installed). The USCG will sample vessels based on a decision matrix to verify compliance with the ballast water treatment program. The primary goal is verifying that treatment technologies capable of meeting the proposed ballast water standard were installed. Two additional goals are verifying the treatment technologies are being used and they continue to meet the ballast water standard after their installation.

The USCG will continue to sample ballast tanks to ensure the ballast water treatment technology is effective (effectiveness monitoring) but the monitoring will depend on both the vessel type and the technology used. Because technologies will be approved on an individual basis, effectiveness monitoring for each particular technology will be evaluated in separate consultations. The tiered consultation is two part, an evaluation of how the technology performs prior to installation and taking water and sediment samples from vessel ballast tanks to ensure the technology is being maintained and functioning properly after installation. The language for the effectiveness technology is included in the FR in 33 CFR 151.2075.

Mitigation Measures

The FR include a section (33 CFR 151.2050) describing appropriate mitigation measures that ships should take in addition to all required ballast water discharge requirements. The mitigation measures are required and will have a positive effect on the environment. The mitigation measures direct vessel owners to avoid ballast water uptake and discharge in marine sanctuaries, marine preserves, marine parks, or coral reefs. Furthermore, the mitigation measures require vessel operators to clean their ballast tanks regularly, discharge only the minimal amount of ballast water necessary, maintain a ballast water management plan, train vessel crew on ballast water treatment procedures, clean the anchor chains, and remove fouling organisms from vessel hulls, pipes, and tanks regularly.

Limitations of Coverage

Certain vessels, including offshore floating platforms, are exempted from this FR by NISA. The vessel exemptions are identified in 33 CFR 151.2015. Crude oil tankers engaged in coastwide trade and vessels operating exclusively within a single captain of the port zone (smaller than a bioregion) are exempted from the ballast water standard, reporting, or record keeping. Additionally, non sea going vessels; sea going vessels of less than 3,000 tons operating across captain of the port zones but not outside of the EEZ; and vessels that take on and discharge ballast water in the same captain of the port zone are exempted from meeting the ballast water standard, but must still report and maintain records as identified above. Exempted vessels are not required to submit a ballast water reporting form to the USCG, but non-exempted vessels are required to report, regardless of whether and how they discharged ballast water.

Implementation Dates

Non-exempt vessels are required to meet the ballast water management standard as early as December 1, 2013 and all others by the first scheduled drydock after January 1, 2016. Vessels constructed on or after December 1, 2013, will be required to meet these standards on delivery, while vessels constructed before December 1, 2013, will need to meet the standard after their first drydocking after January 1, 2014, (vessels with ballast water capacities of 1,500-5,000 m^3) or January 1, 2016 (vessels with ballast water capacities of less than 1,500 m^3 and greater than 5,000 m^3). The USCG will phase out the ballast water exchange requirement and replace it with this numerical standard according to these implementation dates.

Enforcement

Significant civil and criminal penalties are codified in the NISA and the FR that may be enforced against violators of the statute and/or proposed regulations. The various monitoring programs described in this Opinion are part of the USCG enforcement program.

Approach to the Assessment

NMFS approaches its program specific section 7 analyses through a series of steps. The first step identifies those aspects of proposed actions that are likely to have direct and indirect effects on the physical, chemical, and biotic environment. As part of this step, we identify the spatial extent of these direct and indirect effects, including changes in the spatial extent over time. The results of this step represent the action area for the consultation. The second step of our analyses identifies the listed and proposed species and designated or proposed critical habitat 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 primary constituent elements (PCEs), number, age, life stage, and gender of these resources that are likely to be exposed to an action’s effects and the populations or subpopulations those individuals represent. Once we identify the listed and proposed resources that are likely to be exposed to an action’s effects and the nature of that exposure, we examine whether and how those 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 and 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” were listed, which may encompass the biological species, subspecies, or distinct population segments of vertebrate species. Because the continued existence of listed species depends on the fate of the populations that comprise them, the viability (probability of extinction or probability of persistence) of listed and proposed species depends on the viability of the populations that comprise the species. 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 destruction or adverse modification determinations must be based on an action's effects on the conservation values of the essential features of critical habitat.

A programmatic review, however, typically analyzes the general environmental consequences of a broad scope of policy alternatives under consideration by an agency program, in this case, managing a ballast water program to aid in controlling the introduction and spread of non-native species from ship's ballast water. Subsequent consultations that "tier" off of these programmatic consultations, when warranted, analyze the project and site specific effects typical of most consultations. As the proposed rule for the ballast water management program states, there are multiple phases within the program, the first of which is establishing the ballast water standard identified in the FR. Additionally, many ballast water management systems could be used to achieve the ballast water standard, so tiered consultations will evaluate any further impacts that may occur due to chemical, biological, or mechanical treatment of ballast water. All subsequent section 7 consultations conducted by NMFS personnel will be designed to determine whether and to what degree the specific action under review fits within the general pattern identified in the "parent" or national programmatic consultation, and will determine whether the specific action is or is not likely to jeopardize the continued existence of endangered and threatened species or result in the destruction or adverse modification of designated critical habitat.

We begin our programmatic consultations by recognizing that an agency's program normally represents the agency's decision to authorize, fund, or carry out a suite or class of activities that may require subsequent decision-making. When we conduct programmatic examinations of proposals such as the USCG's national ballast water management program, we ask whether or to what degree the Federal action agency (in this case, USCG) has constructed a decision-making process that will consider the information, standards and criteria that NMFS considers during consultations on specific actions. We also ask whether that decision-making process is likely to produce outcomes that will prevent endangered or threatened species and designated critical habitat from being exposed to physical, chemical, or biotic stressors that will directly or indirectly reduce the reproductive success of endangered or threatened individuals, increase the extinction risks of the population(s) those individuals represent, or increase the extinction risks of the species those populations comprise. Specifically, we ask:

1. Has USCG structured the program so they will know or be able to reliably estimate the probable individual and cumulative effects of the action? For example, has USCG structured the program so it will know or be able to reliably estimate the probable number of non-native organisms that will be introduced under the authority of the program? Has USCG structured the program so it will know or be able to reliably estimate the probable location of those introductions that will be authorized by the program?

2. Has USCG structured the program so it will know or be able to reliably estimate the physical, chemical, or biotic stressors that are likely to be produced as a direct or indirect result of the ballast management that will be authorized by the program (that is, the stressors produced by the introduction of non-native species to waters of the US)?

3. Has USCG structured the program so it will know or be able to reliably determine whether or to what degree operators have complied with the conditions, restrictions, or mitigation measures the proposed action requires?

4. Has USCG structured the program so it will know or be able to reliably estimate whether or to what degree specific endangered or threatened species or designated critical habitat are likely to be exposed to (a) chemical, physical, or biological stressors associated with ballast water management, or; (b) to the ecological consequences (habitat alterations or indirect effects to species) as a result of ballast water management in the United States?

5. Has USCG structured the program so it will continuously identify, collect, and analyze information to detect impacts from ballast water management to endangered or threatened species or designated critical habitat resulting from stressors at concentrations, intensities, durations, or frequencies that are known or suspected to produce physical, physiological, behavioral, or ecological responses that have potential individual or cumulative adverse consequences for individual organisms or primary constituent elements of critical habitat?

6. Has USCG structured the program to employ an analytical methodology that considers (a) the status and trends of endangered or threatened species or designated critical habitat; (b) the demographic and ecological status of populations and individuals of those species given their exposure to pre-existing stressors in different drainages and watersheds; (c) the direct and indirect pathways by which endangered or threatened species or designated critical habitat might be exposed to the consequences of organisms carried in ballast to waters of the United States; and (d) the physical, physiological, behavior, sociobiological, and ecological consequences of exposing endangered or threatened species or designated critical habitat to biological stressors from ballast water that are known or suspected to produce physical, physiological, behavioral, or ecological responses, given their pre-existing demographic and ecological condition?

7. Has USCG structured the program so it will be able to limit endangered or threatened species or designated critical habitat from being exposed to stressors resulting from a numerical ballast water standard (a) at concentrations, durations, or frequencies that are potentially harmful to individual listed organisms, populations, or the species, or; (b) to ecological consequences that are potentially harmful to individual listed organisms, populations, the species or primary constituent elements of designated critical habitat? How quickly would USCG be able to implement preventive measures?

Because it is difficult to prevent free-ranging organisms from being affected by anthropogenic stressors once they have been exposed, the most effective management measures are designed to influence the likelihood, duration, and magnitude of the exposure itself. For that reason, our assessment focuses on whether and to what degree the program minimizes endangered and threatened species and designated critical habitat exposure to stressors authorized by the proposed action.

Our assessment focused on whether and to what degree USCG structured the program to minimize endangered or threatened species or critical habitat that has been designated for those species from being exposed to stressors resulting from ballast discharges because such exposures commonly trigger a cascade of events with ultimate consequences difficult to prevent. For example, once individual plants and animals are exposed to a stressor, their responses to the exposure is controlled by the direct and indirect consequences associated with the exposure, their sensitivity to the stressor, other physical, chemical, or biotic stressors that they are exposed to concurrently, their pre-existing physiological state, and their individual fitness.

When individual listed plants or animals experience reductions in fitness, those reductions may reduce the abundance, reproduction rates, or growth rates (or increase variance in one or more of those rates) of the populations those individuals represent (see Stearns 1992). Reductions in one or more 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. On the other hand, when proposed or 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., Brandon 1978, Mills and Beatty 1979, Stearns 1992, Anderson 2000). If we conclude that proposed or listed species are not likely to experience reductions in their fitness, we would conclude our assessment.

If, however, we conclude that proposed or listed species are likely to experience reductions in their fitness, we examine whether the program included sufficient safeguards to ensure that the actions they authorize, fund, or otherwise carry out would not reduce the viability of the populations those individuals represent (typically measured using changes in the populations' abundance, reproduction, spatial structure and connectivity, growth rates, or variance in these measures to make inferences about the population's extinction risks). For those species likely to be adversely affected by the activities conducted under a program, we examine their status and the environment in which the species exists (in this Opinion, the *Environmental Baseline* and *Status of the Species* are examined in the section titled *Action Area*), in detail, as a point of reference for determining if changes in population viability are likely, and if, in turn, any changes in population viability is sufficient to reduce the viability of the species.

In addition to analyzing the ability of the USCG's ballast water management program to insure the action is not likely to jeopardize listed species or destroy or adversely modify their critical habitat, we assess the probable consequences of the specific actions that will be authorized, funded or carried out by the program for endangered and threatened species and critical habitat that has been designated for those species. Specifically:

1. We examine activities that would be authorized by the proposed action.

These analyses summarize (a) establishing a numerical ballast water standard; (b) the geographic distribution of species introductions via ballast water; (c) the frequency of non-native species introductions originating from similar geographic areas; (d) the stressors typically associated with non-native species, and; (e) any additional information on the actions to be authorized.

2. We determine the degree of spatial overlap between ballast water discharges, listed species and designated critical habitat.

These analyses describe spatial overlap and any specific evidence (reports or studies) that particular endangered or threatened species or designated critical habitat have been or are likely to be exposed to the chemical, physical, or biological stressors resulting from those activities. However, this does not represent a detailed exposure analysis: we are merely establishing whether or to what degree endangered or threatened species or designated critical habitat overlap, in space and time (e.g., some non-native populations may establish seasonally but perish during other seasons only to be re-established the following year and their residency may occur while migratory species are not in an area). Given spatial and temporal overlap, we then have reason to ask whether or to what degree USCG's program can insure that these species or critical habitat are not likely to be exposed.

3. We conduct a detailed review of the literature available on the physical, physiological, behavioral, social, and ecological responses of endangered or threatened species or primary constituent elements of critical habitat given exposure to potential stressors, or to the ecological effects (that is, effects resulting from changes in populations of prey, predators, competitors, symbionts, etc.) of those stressors. Rather than discuss the literature for each species, we organized the data using species groups (for example, Pacific salmon; sturgeon; sea turtles; etc.).

4. We summarize the probable consequences of the responses identified in the preceding section for populations of endangered and threatened species and designated critical habitat. Rather than discuss the literature for each species, it is only necessary to discuss the risks of exposing species groups (for example, Pacific salmon; sturgeon; sea turtles; etc.).

In this Opinion, we will present the results of these analyses before we present the results of our review and evaluate the program using the sequence of seven questions we identified previously. We use the results of these combined analyses to determine whether and to what degree USCG structured its program in ways that would protect endangered or threatened species or critical habitat that has been designated for those species.

Evidence Available for this Consultation

To conduct our analyses we considered the information contained in the USCG’s letter requesting initiation of consultation, the final PEIS, the FR, 33 CFR 151, NANPCA, NISA, and monitoring data from previous ballast water rules. We supplemented this information using electronic searches of literature published in English or with English abstracts using research platforms in the *Online Computer Library Center’s First Search, CSA Illumina,* and *ISI Web of Science.* These platforms allow us to cross-search multiple databases for journals, open access resources, books, proceedings, Web sites, doctoral dissertations and master’s theses for literature on the biological, ecological, and medical sciences. Particular databases we searched for this consultation included *BasicBiosis, Dissertations, ArticleFirst, Proceedings, Aquatic Sciences and Fisheries Abstracts, BioOne Abstracts & Indexes, and Water Resources Abstracts*. Some of the databases provide access to documents published from the 1960s through present, although references for many scientific journals contained in these databases only date back to the 1970s or later. Through these databases we accessed the major journals dealing with the biology, ecology, distribution, status, and trends of the threatened and endangered species considered in this Opinion, and the impacts of invasive species on freshwater, estuarine, and marine ecosystems.

For our literature searches, we used paired combinations of the keywords: ballast water, discharge, invasive species, habitat, non-indigenous, and many others. We acquired references that, based on a reading of their titles and abstracts, appeared to comply with our keywords. If a reference's title appeared relevant, we acquired and reviewed the reference. We supplemented our electronic searches by searching the literature cited sections and bibliographies of references we retrieved electronically to identify additional papers that had not been captured in our electronic searches.

Collectively, this information provided the basis for our determination as to whether and to what degree listed resources under our jurisdiction are likely to be effected by the USCG's ballast water management program, and whether and to what degree the USCG can insure that its ballast water management program is not likely to jeopardize the continued existence of proposed, endangered, or threatened species or result in the destruction or adverse modification of proposed or designated critical habitat.

USCG Decision Structure

Under the ballast water management program, one of the early and most important actions taken by the USCG is the review of available technologies to understand the best available treatment technologies at the time of the proposal. Based on the feasibility of the technologies, the ballast water standard is established. The USCG also conducts annual monitoring and reporting as well as a three-year periodic review. All of those reviews are intended to provide for flexibility and adaptive management within the USCG ballast water management program.

Figure 1 depicts a simplified model of the USCG ballast water management program, as NMFS understands it. Establishing a national ballast water standard, an action that merits consultation, represents only one part of the overall program and decision making process in regulating ballast water discharges. During this consultation, we evaluated the proposed ballast water standard and the USCG's decision-making process for when and how to establish a more stringent standard if appropriate. We reviewed the monitoring plan and practicability review that the USCG analyzes and considers to reach conclusions as to whether listed species would be affected by numerical ballast water standards. We evaluated this information to determine whether and to what degree the USCG's decision-making process insures that any activities it authorizes, funds, or carries out are not likely to, individually or cumulatively, jeopardize the continued existence of proposed or listed species or result in the destruction or adverse modification of proposed or designated critical habitat.

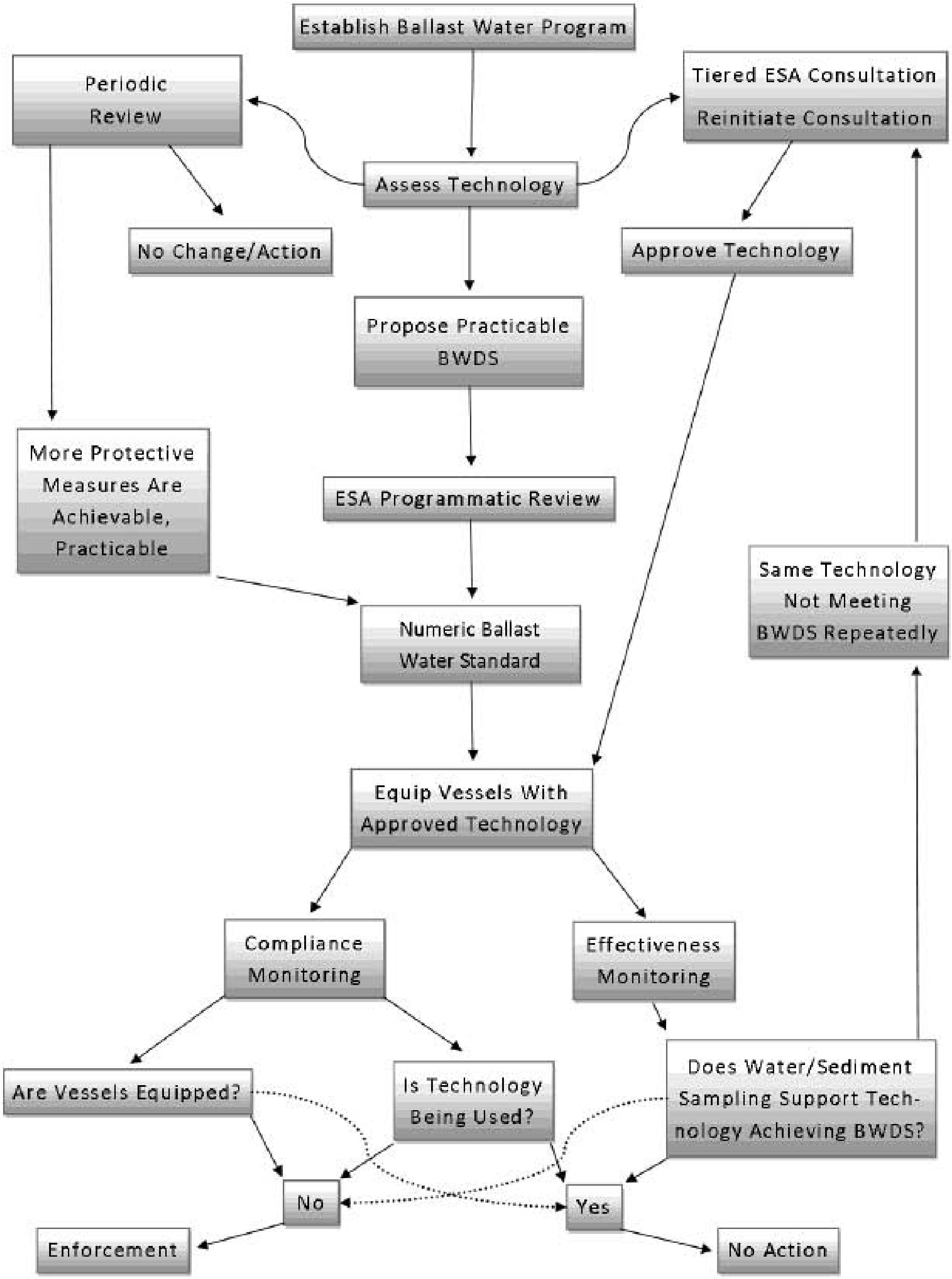

Figure 1. Diagram of USCG Ballast Water Management Program decision making process.

Application of this Approach in this Consultation

In this consultation, we evaluated the USCG's proposed ballast water management program, including the numerical standards, and whether the USCG has insured that any action authorized, funded, or carried out under this ballast water management program is not likely to jeopardize the continued existence of any proposed, endangered, or threatened species, or result in the destruction or adverse modification of proposed or designated critical habitat. We began our analysis of the ballast water management program by exploring the USCG's preferred alternative in relation to the ballast water management program. Specifically, we reviewed the rationale used to identify the preferred alternative, including all substantive decision criteria that influenced the USCG's decision to establish that specific ballast water standard. After evaluating the preferred alternative, we assessed the ballast water program's monitoring programs: three-year periodic reviews, compliance monitoring, effectiveness monitoring, and environmental monitoring. The results of those monitoring actions will feed back into the decision making process for future potential numerical standards.

In addition, we asked: how does the USCG determine a numerical standard, how frequently does the USCG conduct program reviews, what is the criteria that triggers an adaptive management adjustment to ballast water standards, which ships are not bound by the USCG's ballast water regulations, is the USCG able to influence improved ballast water standards through incentive programs or research funding, is the USCG able to determine when a ship has installed an acceptable technology, is the USCG able to determine the technology still meets the standard after it has been installed, is the USCG able to determine if the technology is able to meet the standard after having been used for years, is the USCG able to determine whether implementation of the numerical standard has effectively reduced or prevented new invasive species, and what is the ability of the USCG to monitor newly introduced species?

Through the course of this consultation we learned that the decision to identify the proposed ballast water standard was dependent on the "maximum extent practicable" mandate of NISA. This decision is informed by vessel and human safety, water quality considerations, NANCPA, NISA, economic impacts to commerce and society, and the proven ability of ballast water treatment techniques to achieve the proposed ballast water standards.

This consultation will evaluate the ballast water management program by describing the likelihood of listed species or their critical habitat being exposed to non-native species, introduced through the proposed ballast water standard. To best analyze the likelihood of exposure, we will conduct a primary analysis of several large ports in the US with species compositions that encompass all listed functional groups. As a secondary analysis of exposure, we will conduct specific analyses of ports with listed species not identified in the primary analysis. To accomplish this and to understand the responses of listed species, we will consider both listed species and introduced species as functional groups, such as salmonids, pinnipeds, and sea turtles or diseases, predators, and competitors, respectively. Once we have established the likely rate of exposure to listed species, using the response analysis, we will determine the proportion of invasive species that would likely be harmful. Using the information from the exposure and response analyses, we can determine the risks ballast water discharge at the proposed standards may pose to listed species and their critical habitat in order to reach a conclusion about the USCG's program and its ability to insure the USCG is not likely to jeopardize any listed species or destroy/adversely modify their critical habitat.

During this consultation, NMFS considered the various ballast water discharge standards proposed by different states and authorized, funded, or carried out by other Federal agencies under different legal mandates. NMFS did not include those other actions in the analysis, but instead evaluated: whether the approved state standards are achievable, the outcomes of legal action against those enacted state standards, and the approval process for EPA vessel discharge permit program in order to understand how the USCG program established the numerical standards and whether there are other state or Federal laws that influence the ballast water management program.

There are other consultations that may necessarily depend on the outcome of this consultation and be tiered from this consultation in the future (see Figure 1). If the USCG determines there may be an effect as a result of new technologies being developed to achieve the proposed numerical standard, NMFS will evaluate the effects of those technologies on proposed and listed resources. However, future consultations with EPA on its vessel discharge program or any consultation on state standards with a federal nexus requiring section 7 consultation will be separate actions not considered under USCG's ballast water management program. Even if the ballast water program complies with the requirements of section 7(a)(2) of the Endangered Species Act (that is, the requirement to insure that their actions are not likely to jeopardize the continued existence of endangered or threatened species or result in the destruction or adverse modification of critical habitat that has been designated for those species), the reinitiation criteria at the end of this Opinion could require further analysis of this program.

We examined the USCG ballast water management program to see if it contains features that minimize the exposure of proposed, endangered, or threatened species, or their proposed or designated critical habitat to harmful non-native species. When exposure is not avoided, we broadly characterized the uptake and discharge of ballast water at US ports with specific emphasis on San Francisco Bay, Puget Sound, and Chesapeake Bay to describe the risk of listed resources being exposed to uptake or discharge of ballast water at all US ports. If, based on this information, we expect that listed resources are not likely to be exposed to non-native introduced species, then we conclude that the action would have "no effect" on those listed resources. If, based on this information, we determined that listed individuals may be exposed to uptake of ballast water or non-native introduced species, but (a) the probability of exposure is so small that it would not be reasonable to expect exposure to occur, (b) there is no possibility or only a very small possibility that the individual listed species would respond to exposure, (c) there is no possibility or only a small probability of a negative response even if there is a response to uptake in ballast water tanks or to introduced non-native species, or (d) there is no possibility or only a small probability that the individual would experience a reduction in individual performance (or fitness), then we concluded that the USCG's action is "not likely to adversely affect" those proposed or listed resources.

If proposed or listed resources or their critical habitats may be lethally or sub-lethally affected by actions the USCG's ballast water management program authorizes, funds, or otherwise carries out, NMFS examines if the program includes sufficient safeguards to insure that the incidental take of individuals does not occur in a manner that reduces the viability of the populations those individuals represent (typically measured using changes in the populations' abundance, reproduction, spatial structure and connectivity, growth rates, or variance in these measures to make inferences about the population's extinction risks). Given their status and the environment in which the species exist, are those species likely to be adversely affected by the activities conducted under the proposed action also likely to suffer changes in population viability that would be sufficient to reduce the viability of the species those populations comprise?

ACTION AREA

The section 7 implementing regulations define the "Action Area" of a Federal action as all areas to be affected, directly or indirectly, and not merely the immediate area involved in the action (50 CFR 402.02). This Opinion assesses the consequences of the USCG's establishment of ballast water standards in all waters of the US. The USCG's proposed rule establishes the action area as all navigable waters within the US extending offshore to the outer edge of the EEZ, 200 miles offshore of the US coastline except where this region overlaps with the EEZ of another nation (Mexico, Canada, Russia, Cuba, the Federation of Micronesian States, and The Bahamas). Furthermore, because vessels entering the Great Lakes are still required to conduct ballast water exchange beyond the EEZ, ballast water intake from areas beyond the EEZ along the East Coast of America are also part of this action area.

Status of Proposed and Listed Resources

The USCG ballast water management program may affect the threatened and endangered species and critical habitat listed in Table 1. We excluded foreign species because the USCG has no jurisdiction to regulate ballast water in foreign countries.

Table 1. Proposed and listed resources in the action area. Asterisks denote critical habitat in the action area. Double asterisks denote proposed critical habitat in the action area.

Common name (Distinct population segment, evolutionarily significant unit, or subspecies)	***Scientific name***	***Status***
Cetaceans		
Blue whale	*Balaenoptera musculus*	Endangered
Bowhead whale	*Balaena mysticetes*	Endangered
Fin whale	*Balaenoptera physalus*	Endangered
Humpback whale	*Megaptera novaeangliae*	Endangered
Killer whale (Southern Resident*)	*Orcinus orca*	Endangered
North Atlantic right whale*	*Eubalaena glacialis*	Endangered
North Pacific right whale*	*Eubalaena japonica*	Endangered
Sei whale	*Balaenoptera borealis*	Endangered
Sperm whale	*Physeter macrocephalus*	Endangered
Beluga whale (Cook Inlet)**	*Delphinapterus leucas*	Endangered
Pinnipeds		
Guadalupe fur seal	*Arctocephalus townsendi*	Threatened
Hawaiian monk seal*, **	*Monachus schauinslandi*	Endangered
Steller sea lion (Eastern*)	*Eumetopias jubatus*	Threatened
Steller sea lion (Western*)		Endangered
Marine Turtles		

Common name (Distinct population segment, evolutionarily significant unit, or subspecies)	***Scientific name***	***Status***
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 (northwestern Atlantic)	*Caretta caretta*	Threatened
Loggerhead sea turtle (North Pacific)		Endangered
Loggerhead sea turtle (South Pacific)		Threatened
Olive ridley sea turtle (Mexico's Pacific coast breeding colonies)	*Lepidochelys olivacea*	Endangered
Olive ridley sea turtle (All other areas)		Threatened
Anadromous Fishes		
Atlantic salmon (Gulf of Maine*)	*Salmo salar*	Endangered
Chinook salmon (California Coastal*)	*Oncorhynchus tschawytscha*	Threatened
Chinook salmon (Central Valley Spring-run*)		Threatened
Chinook salmon (Lower Columbia River*)		Threatened
Chinook salmon (Upper Columbia River Spring-run*)		Endangered
Chinook salmon (Puget Sound*)		Threatened
Chinook salmon (Sacramento River Winter-run*)		Endangered
Chinook salmon (Snake River Fall-run*)		Threatened
Chinook salmon (Snake River Spring/Summer-run*)		Threatened
Chinook salmon (Upper Willamette River*)		Threatened
Chum salmon (Columbia River*)	*Oncorhynchus keta*	Threatened
Chum salmon (Hood Canal Summer-run*)		Threatened
Coho salmon (Central California Coast*)	*Oncorhynchus kisutch*	Endangered
Coho salmon (Lower Columbia River)		Threatened
Coho salmon (Southern Oregon & Northern California Coast*)		Threatened
Coho salmon (Oregon Coast*)		
Green sturgeon (Southern*)	*Acipenser medirostris*	Threatened
Shortnose sturgeon	*Acipenser brevirostrum*	Endangered
Smalltooth sawfish*	*Pristis pectinata*	Endangered
Bocaccio (Georgia Basin)	*Sebastes paucispinis*	Endangered
Yelloweye rockfish (Georgia Basin)	*Sebastes pinniger*	Threatened
Canary rockfish (Georgia Basin)	*Sebastes ruberrimus*	Threatened
Pacific eulachon**	*Tha;eichthys 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
Atlantic sturgeon (Gulf of Maine)	*Asipenser oxyrincus*	Threatened
Atlantic sturgeon (New York Bight)		Endangered
Atlantic sturgeon (Chesapeake Bay)		Endangered
Atlantic sturgeon (Carolina)		Endangered
Atlantic sturgeon (South Atlantic)		Endangered

Common name (Distinct population segment, evolutionarily significant unit, or subspecies)	*Scientific name*	*Status*
Marine Invertebrates		
Elkhorn coral*	*Acropora palmata*	Threatened
Staghorn coral*	*Acropora cervicornis*	Threatened
White abalone	*Haliotis sorenseni*	Endangered
Black abalone*	*Haliotis cracherodii*	Endangered
Marine Plants		
Johnson's seagrass*	*Halophilia johnsonii*	Threatened
Proposed for listing		
False killer whale (Hawaiian 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

We do not expect exposure of several listed or proposed species or critical habitats that occur exclusively (or nearly so) in offshore or deepwater marine environments. Based upon review of available literature detailed further in our *Response analysis*, we discount the possibility that invasive species established from ballast water discharged in US waters (primarily US ports, which occupy nearshore or inland waters) would spread to offshore habitats. Inshore, estuarine, and freshwater habitats are fundamentally different in ecological terms and we discount the ability of an invasive species to establish in offshore regions as well as one of these areas where the large majority of ballast water discharge would occur. Based upon this, we discount the exposure of blue, fin, sei, sperm, North Pacific right, and Hawaiian insular false killer whales. For the same reasons, we also discount exposure to North Pacific right whale critical habitat.

Several listed or proposed resources also occur exclusively (except for extralimital occurrences) in areas where ballast water discharge is so limited that the probability of establishment of non-native species and subsequent exposure of listed resource is discountable. For some species, the isolated and/or offshore nature of the species was also a consideration. These include bowhead whales, Beringia DPS bearded seals, and Arctic DPS ringed seals.

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

Cetaceans

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 (Katona and Beard 1990, Smith *et al.* 1999b, Boye *et al.* 2010). 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 (Winn *et al.* 1975, Balcomb III and Nichols 1982, Whitehead and Moore 1982). The largest breeding aggregations occur off the Greater Antilles where humpback whales from all North Atlantic feeding areas have been photo-identified (Katona and Beard 1990, Clapham *et al.* 1993, Mattila *et al.* 1994, Palsbøll *et al.* 1997, Smith *et al.* 1999b, 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 (Reiner *et al.* 1996, Reeves *et al.* 2002, Weir 2007, Cerchio *et al.* 2010). Accessory and historical aggregations also occur in the eastern Caribbean. 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 (Nemoto 1957, Tomilin 1967, Johnson and Wolman 1984). 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 (Eldredge 1991, Darling and Mori 1993, Rice 1998a, Eldredge 2003). During summer, whales from this population migrate to the Kuril Islands, Bering Sea, Aleutian Islands, Kodiak, Southeast Alaska, and British Columbia to feed.

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 (Waite *et al.* 1999, Urban *et al.* 2000, Calambokidis *et al.* 2001a, Witteveen *et al.* 2004, Calambokidis *et al.* 2009). 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 (Consiglieri *et al.* 1982, Baker *et al.* 1985, Straley 1990, Waite *et al.* 1999, Stafford *et al.* 2007, Dahlheim *et al.* 2008). During this time, feeding aggregations are found throughout the Kodiak Archipelago, although pelagic areas may also be important foraging areas (MMC 2002, Baraff *et al.* 2005). 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.* 2010b).

Southern Hemisphere. Eight proposed stocks, or populations, of humpback whales occur in waters off Antarctica (Figure 2). 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 heterogenous from individuals in other breeding areas, but appear to maintain a genetioc 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 50º 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).

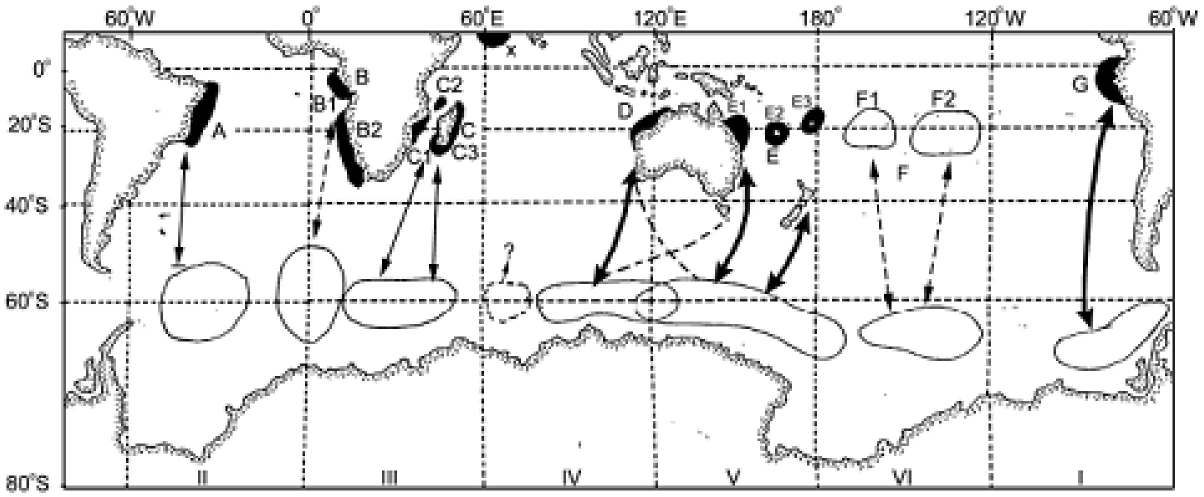

Figure 2. Southern Hemisphere humpback stocks (populations)(IWC 2005).

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 (Glockner-Ferrari and Ferrari 1985 as cited in NMFS 2005b, Clapham and Mayo 1987, 1990, 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).

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 (Jurasz and Jurasz 1979, Hain *et al.* 1982, Weinrich *et al.* 1992, Hain *et al.* 1995, Witteveen *et al.* 2011). The principal fish prey in the western North Atlantic are sand lance, herring, and capelin (Kenney *et al.* 1985b). There is good evidence of some territoriality on feeding and calving areas (Tyack 1981, Clapham 1994, Clapham 1996). 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 two to four individuals, although larger groups are seen near Shuyak and Sitkalidak islands and groups of 20 or more have been documented (Wynne *et al.* 2005).

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 2 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 (Katona and Beard 1990, Barlow 1997, Smith *et al.* 1999b, Waring *et al.* 2001). The rate of increase varies from 3.2-9.4%, with rates of increase slowing over the past two decades (Katona and Beard 1990, Barlow 1997, 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.* (2009) 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 (Rice 1978, Calambokidis *et al.* 1997). It is estimated that 15,000 humpback whales resided in the North Pacific in 1905 (Rice 1978). 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, Darling and Morowitz 1986, Baker and Herman 1987, Calambokidis *et al.* 1997). 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).

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

Region	Population, stock, or study area	Pre-exploitation estimate	95% C.I.	Current estimate	95% C.I.	Source
Global	--	1,000,000	--	--	--	(Roman and Palumbi 2003)
North Atlantic	Basinwide	240,000	156,000-401,000*	11,570	10,005-13,135*	(Roman and Palumbi 2003) (Stevick *et al.* 2001) *in* (Waring *et al.* 2004b)
	Basinwide - Females	--	--	2,804	1,776-4,463	(Palsbøll *et al.* 1997)
	Basinwide - Males	--	--	4,894	3,374-7,123	(Palsbøll *et al.* 1997)
	Western North Atlantic from Davis Strait, Iceland to the West Indies	>4,685*	--	--	--	*circa 1865; (Mitchell and Reeves 1983)
	NMFS - Gulf of Maine stock	--	--	845	CV=0.55	(NMFS 2008b)
	NMFS - Gulf of Maine stock, including a portion of Scotian Shelf	--	--	902	177-1,627*	(Clapham *et al.* 2003)
	Northeast Atlantic - Barents and Norwegian Seas	--	--	889	331-1,447*	(Øien 2001) *in* (Waring *et al.* 2004b)
North Pacific	Basinwide	15,000	--	6,000-8,000	--	(Calambokidis *et al.* 1997)
	NMFS - Western North Pacific stock	--	--	394	329-459*	(Angliss and Allen 2007)
	NMFS - Central North Pacific stock	--	--	4,005	3,259-4,751*	(Angliss and Allen 2007)
	NMFS - Eastern North Pacific stock	--	--	1,391	1,331-1,451*	(Carretta *et al.* 2008)
Indian Ocean	Arabian Sea	--	--	56	35-255	Minton *et al.* (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 (Slijper *et al.* 1964, Wray and Martin. 1983, Reeves *et al.* 1991, Mikhalev 2000, Minton *et al.* 2008).

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). Modelling 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 survivial 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 (Perkins and Beamish 1979, Lien 1994). Along the Atlantic coast of the US 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.* 2005c, Nelson *et al.* 2007c). 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).

More humpback whales are killed in collisions with ships than any other whale species except fin whales (Jensen and Silber 2003). Of 123 humpback whales that stranded along the Atlantic coast of the US 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 US and the Maritime Provinces of Canada (Cole *et al.* 2005c, Nelson *et al.* 2007c). 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 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.* 1997b). 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.

North Atlantic right whale

Description of the species. All North Atlantic right whales compose a single population. Although not all individuals undergo the same migratory pattern, no subpopulation structuring has been identified.

Distribution. Right whales occur in sub-polar to temperate waters in all major ocean basins in the world, with a clear migratory pattern of high latitudes in summer and lower latitudes in winter (Cummings 1985, Rice 1998b, Perry *et al.* 1999). The historical range of North Atlantic right whales extended as far south as Florida and northwestern Africa, and as far north as Labrador, southern Greenland, Iceland, and Norway (Reeves *et al.* 1978, Cummings 1985, Rice 1998b). Most sightings in the western North Atlantic are concentrated within five primary habitats or high-use areas: coastal waters of the southeastern US, Cape Cod and Massachusetts Bays, the Great South Channel, the Bay of Fundy, and the Scotian Shelf (Winn *et al.* 1986). In 1994, the first three of these areas were designated as critical habitat for the North Atlantic right whale.

North Atlantic right whales have been observed from the mid-Atlantic Bight northward through the Gulf of Maine year-round, but are primarily found along the northeast US during summer and Florida during winter, with migratory routes in between. In New England, peak abundance of North Atlantic right whales in feeding areas occurs in Cape Cod Bay beginning in late winter. In early spring (Late February to April), peak North Atlantic right whale abundance occurs in Jordan and Wilkinson basins to the Great South Channel (Kenney *et al.* 1995, Nichols *et al.* 2008, Pace III and Merrick 2008). In late June and July, North Atlantic right whale distribution gradually shifts to the northern edge of Georges Bank. In late summer (August) and fall, much of the population is found in waters in the Bay of Fundy, the western Gulf of Maine and around Roseway Basin (Winn *et al.* 1986, Kenney *et al.* 1995, Kenney *et al.* 2001, Pace III and Merrick 2008). However, year-to-year variation in space and time are known and likely result from patchy prey distribution (Nichols *et al.* 2008). Variation in the abundance and development of suitable food patches appears to modify the general patterns of movement by reducing peak numbers, stay durations and specific locales (Brown *et al.* 2001, Kenney 2001). In particular, large changes in the typical pattern of food abundance will dramatically change the general pattern of North Atlantic right whale habitat use (Kenney 2001).

Migration and movement. North Atlantic right whales exhibit extensive migratory patterns, traveling along the eastern seaboard of the US and Canada between calving grounds off Georgia and Florida to northern feeding areas off of the northeast US and Canada in March/April and the reverse direction in November/December. The longest tracking of a North Atlantic right whale was a migration of 1,200 miles in 23 days the Bay of Fundy to Georgia (Mate and Baumgartner 2001). Migrations are typically within 30 nautical miles of the coastline and in waters less than 160 feet deep. Although this pattern is well-known, most of the population, particularly the males and non-pregnant females, is not found in the calving area and may not follow this pattern. It is unknown where the majority of the non-calving population spends the winter.

There have been a few recent sightings of North Atlantic right whales far offshore, including those from Dutch ships indicating some individuals occur between 40° and 50°N, in waters influenced by the North Atlantic Current (the broad, eastward-flowing extension of the Gulf Stream). Right whales have been sighted offshore (greater than 30 miles) during surveys flown off the coast of northeastern Florida and southeastern Georgia from 1996 to 2001. These include three sightings in 1996, one in 1997, 13 in 1998, six in 1999, 11 in 2000, and six in 2001 (within each year, some were repeat sightings). Mate *et al.* (1997) recorded radio-tagged animals making extensive movements from the Gulf of Maine into deeper waters off the continental shelf (Mate *et al.* 1997). The frequency with which North Atlantic right whales occur in offshore waters in the southeastern US remains unclear. Occasionally, individuals are observed in distant locations, including the Gulf of Mexico, Bermuda, the Gulf of St. Lawrence, Newfoundland, Greenland, Iceland, and northern Norway (an area known as a historical North Atlantic right whale feeding area Smith *et al.* 2006). The Norwegian sighting (September 1992) represents one of only two sightings this century of a right whale in Norwegian waters, and the first since 1926. Together, these long-range matches indicate an extended range for at least some individuals and perhaps the existence of important habitat areas not presently well described.

Reproduction and demography. Data through the 1990s suggests that mean calving interval increased since 1992 from 3.67 years to more than five years, a significant trend that hampers North Atlantic right whale recovery (Best *et al.* 2001a, Kraus *et al.* 2007). This reproductive rate was approximately half that reported from studied populations of southern right whales (Best *et al.* 2001b). This has been attributed to several possible causes, including higher abortion or perinatal losses (Browning *et al.* 2009). An analysis of the age structure of North Atlantic right whales suggests that the population contains a smaller proportion of juvenile whales than expected, which may reflect lowered recruitment and/or high juvenile mortality (Hamilton *et al.* 1998, Best *et al.* 2001a). In addition, it is possible that the apparently low reproductive rate is due in part to unstable age structure or to reproductive senescence on the part of some females. However, knowledge on either factor is poor. Even though investment in calves is high for North Atlantic right whales, an incident of calf exchange (probably accidentally and soon after birth) and subsequent adoption through weaning has been found (Frasier *et al.* 2010). Although North Atlantic right whales historically separated from their calves within one year, a shift appears to have taken place around 2001 where mothers (particularly less experienced mothers) return to wintering grounds with their yearling at a much greater frequency (71% overall)(Hamilton and Cooper. 2010). The significance of this change is unknown.

Habitat. Available evidence from North Atlantic right whale foraging and habitat studies shows that North Atlantic right whales focus foraging activities where physical oceanographic features such as water depth, current, and mixing fronts combine to concentrate copepods (Wishner *et al.* 1988, Murison and Gaskin 1989, Mayo and Marx 1990, Baumgartner *et al.* 2003).

Feeding. North Atlantic right whales fast during the winter and feed during the summer, although some may opportunistically feed during migration. North Atlantic right whales use their baleen to sieve copepods from dense patches, found in highly variable and spatially unpredictable locations in the Bay of Fundy, Roseway Basin, Cape Cod Bay, the Great South Channel, and other areas off northern US and Canada (Pendleton *et al.* 2009). The primary prey of North Atlantic right whales is zooplankton, especially shrimp-like copepods such as *Calanus* (Kenney *et al.* 1985a, Beardsley *et al.* 1996). North Atlantic right whales feed largely by skimming these prey from the ocean surface (Pivorunas 1979, Mayo and Marx 1990), but may feed anywhere in the water column (Watkins and Schevill 1976, 1979, Goodyear 1993, Winn *et al.* 1995). Feeding behavior has only been observed in northern areas and not on calving grounds or during migration (Kraus *et al.* 1993).

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 North Atlantic and North Pacific right whales are separate species. Following a comprehensive status review, NMFS concluded that North Atlantic and North Pacific right whales are separate species. In March 2008, NMFS published a final rule listing North Pacific and North Atlantic right whales as separate species (73 FR 12024).

North Atlantic right whales were formerly abundant, with an estimated 5,500 individuals present in the 16th century throughout the North Atlantic (Reeves 2001, Reeves *et al.* 2007). A review of the photo-id recapture database in June 2006, indicated that only 313 individually recognized North Atlantic right whales were observed during 2001. This represents a nearly complete census, and the estimated minimum population size. However, no estimate of abundance with an associated coefficient of variation has been calculated for the population. The population growth rate reported for the period 1986 to 1992 by Knowlton *et al.* (1994) was 2.5%, suggesting the stock was showing signs of slow recovery. However, work by Caswell *et al.* (1999) suggested that crude survival probability declined from about 0.99 in the early 1980's to about 0.94 in the late 1990s. Additional work conducted in 1999 showed that survival had indeed declined in the 1990s, particularly for adult females (Best *et al.* 2001a). Another workshop in September 2002 further confirmed the decline in this population (Clapham 2002).

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

Other natural factors influencing right whale recovery are possible, but unquantified. 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. Several human activities are known to threaten North Atlantic right whales: whaling, commercial fishing, shipping, and environmental contaminants. Historically, whaling represented the greatest threat to every population of right whales and was ultimately responsible for listing right whales as an endangered species. As its legacy, whaling reduced North Atlantic right whales to about 300 individuals in the western North Atlantic Ocean; the number of North Atlantic right whales in the eastern North Atlantic Ocean is probably much smaller, although we cannot estimate the size of that population from the data available.

Of the current threats to North Atlantic right whales, entanglement in commercial fishing gear and ship strikes pose the greatest threats. Along the Atlantic coast of the US and the Maritime Provinces of Canada, there were 43 reports of North Atlantic right whales entangled in fishing gear between 1999 and 2005 (Cole et al. 2005a). Of the 39 reports that NMFS could confirm, North Atlantic right whales were injured in five of the entanglements and killed in four entanglements. Recent efforts to disentangle right whales have met with success (Anonmyous. 2009b). In the same region, there were 18 reports of North Atlantic right whales being struck by vessels between 1999 and 2005 (Cole *et al.* 2005a, Nelson *et al.* 2007a). Of the 17 reports that NMFS could confirm, right whales were injured in two of the ship strikes and killed in nine. Present recommendations for slower vessel speeds in the Bay of Fundy appear to be largely ignored (Vanderlaan *et al.* 2008b). Proposed rules for seasonal (June through December) slowing of vessel traffic to 10 knots or changing shipping lanes by less than one nautical mile to avoid the greatest concentrations of right whales are predicted to be capable of reducing ship strike mortality by 62% in the Bay of Fundy region.

Concern also exists over climate change and its effect on the ability of North Atlantic right whales to recover (Greene *et al.* 2003b). Specifically, the variations in oceanography resulting from current shifts and water temperatures can significantly affect the occurrence of the North Atlantic right whale's primary food, copepod crustaceans. If climate changes such that current feeding areas cannot sustain North Atlantic right whales, the population may have to shift to reflect changes in prey distribution, pursue other prey types, or face prey shortage. Changes in calving intervals with sea surface temperature have already been documented for southern right whales (Leaper *et al.* 2006).

North Atlantic right whales, as with many marine mammals, are exposed to numerous toxins in their environment, many of which are introduced by humans. Levels of chromium in North Atlantic right whale tissues are sufficient to be mutagenic and cause cell death in lung, skin, or testicular cells and are a concern for North Atlantic right whale recovery (Wise *et al.* 2008, Chen *et al.* 2009). The organochlorines DDT, DDE, PCBs, dieldrin, chlordane, HCB, and heptachlor epoxide have been isolated from blubber samples and reported concentrations may underestimate actual levels (Woodley *et al.* 1991). Mean PCB levels in North Atlantic right whales are greater than any other baleen whale species thus far measured, although less than one-quarter of the levels measured in harbor porpoises (Van Scheppingen *et al.* 1996, Gauthier *et al.* 1997a). Organochlorines and pesticides, although variable in concentration by season, do not appear to currently threaten North Atlantic right whale health and recovery (Weisbrod *et al.* 2000). Flame retardants such as PBDEs (known to be carcinogenic) have also been measured in North Atlantic right whales (Montie *et al.* 2010).

Critical habitat. Critical habitat is designated for right whales in the North Atlantic. NMFS designated three areas in June 1994 as critical habitat for *Eubalaena glacialis* for feeding and calving (59 FR 28805). The critical habitats for feeding cover portions of the Great South Channel (east of Cape Cod), Massachusetts Bay and Cape Cod Bay, and Stellwagen Bank. Northern critical habitat was designated because of the concentration of right whales that feed in the area, apparently associated with complex oceanographic features that drive prey density and distribution. This area has come under considerable scrutiny within the past few years because of the concern over ship strikes in this area. Boston serves as a major port facility and vessels transiting to and from the port cross critical habitat where North Atlantic right whale mortality occurs. Shipping traffic has generally increased in the recent past and could be considered to degrade the habitat due to the additional mortality and injury risk now present in the area. Although voluntary regulations are in place, these are frequently ignored and mandatory regulations are under consideration. The southern critical habitats are along Georgia and northeastern Florida coasts (waters from the coast out 15 nautical miles between the latitudes of 31°15' N and 30°15' N and from the coast out five nautical miles between 30°15' N and 28°00' N). Southern critical habitat is designated to protected calving and breeding grounds for North Atlantic right whales, which generally calve and breed in shallow coastal waters. This critical habitat has generally fared better than northern critical habitat and significant degradation has not been clearly identified.

Southern resident killer whale

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

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

Movement and habitat. Southern Residents are highly mobile and can travel up to 100 miles per day (Erickson 1978, Baird 2000). Members of K and L pods once traveled a straightline 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)(Heimlich-Boran 1988, Felleman *et al.* 1991, 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 2005g). 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 (Heimlich-Boran 1988, Felleman *et al.* 1991, Olson 1998, Ford *et al.* 2000). 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, 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 (Scheffer and Slipp 1948, Ford *et al.* 1998, Ford *et al.* 2000, Saulitis *et al.* 2000, Ford and Ellis 2005, 2006). Killer whales show a strong preference for Fraser River Chinook salmon (78% of identified prey) during late spring to fall (Hanson *et al.* 2005, Ford and Ellis 2006, Hanson *et al.* 2010b). 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.* 2009). 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 (Olesiuk *et al.* 1990a, Matkin *et al.* 2003). 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, Perrin and Reilly 1984, Duffield and Miller 1988, Olesiuk *et al.* 1990a). Most mating is believed to occur from May to October (Nishiwaki 1972, Olesiuk *et al.* 1990a, Matkin *et al.* 1997). 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 (Nishiwaki and Handa 1958, Olesiuk *et al.* 1990a, Clark *et al.* 2000, Ford 2002). Mothers and offspring maintain highly-stable, life-long social bonds and this natal relationship is the basis for a matrilineal social structure (Bigg *et al.* 1990, Baird 2000, Ford *et al.* 2000). Some females may reach 90 years of age (Olesiuk *et al.* 1990a).

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.* 2002a). 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 2005g). 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 2008).

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 (Ford *et al.* 2000, Carretta *et al.* 2005). The population grew to 90 whales in 2006, although it declined to 87 in 2007 (NMFS 2008f). The most recent population abundance estimate of 87 Southern Residents consists of 25 whales in J pod, 19 whales in K pod, and 43 whales in L pod (NMFS 2008f).

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

Anthropogenic threats. Numerous threats to the continued survival of Southern Resident killer whales have been identified (NMFS 2008f). Many of these are human in origin. The primary prey of killer whales, salmon, has been severely reduced due to habitat loss and overfishing of salmon along the West Coast (NRC 1996c, Slaney *et al.* 1996, Gregory and Bisson 1997, Lichatowich 1999, Lackey 2003, Pess *et al.* 2003, Schoonmaker *et al.* 2003). 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.* 2009). 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.* 2009). Presently, the greatest contaminant threats are organochlorines, which include PCBs, pesticides, dioxins, furans, other industrial products, and the popularized chemical DDT (Ross *et al.* 2000a, CBD 2001b, Krahn *et al.* 2002a, Cullon *et al.* 2009, Krahn *et al.* 2009). 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.* 2009). 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 asa 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 US naval vessel sonar activities, ship strike, and heavy and continuous presence by whale-watching vessels. In 2005, a US 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, Gordon and Moscrop 1996, Erbe 2002a, Williams *et al.* 2002a, Williams *et al.* 2002c, NMFS 2008f, Holt *et al.* 2009). Commercial whale-watching in the region focuses primarily on Southern Resident killer whales and has increased dramatically in the recent years (Osborne *et al.* 1999, Baird 2001a, Erbe 2002a, MMMP 2002a, Koski 2004, 2006b, 2007a). 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 (Kruse 1991, Kriete 2002, Williams *et al.* 2002a, Williams *et al.* 2002c, Foote *et al.* 2004a, Bain *et al.* 2006b, Bain *et al.* 2006c, NMFS 2008f, Wiley *et al.* 2008, Noren *et al.* 2009).

Critical habitat. Critical habitat for the DPS of Southern Resident killer whales was designated on November 29, 2006 (71 FR 69054). Three specific areas were designated; (1) the Summer

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

Cook Inlet beluga whale

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

Distribution. Beluga whales are observed year-round in Cook Inlet although less is known about their winter movements than summer movements (Hobbs *et al.* 2008). Data from satellite tagging studies suggest that movements of Cook Inlet beluga whales during summer months are short and largely focused around river estuaries and inlets (e.g., Chickaloon Bay, Turnagain Arm, Susitna River, and Knik Arm in the upper inlet and in many cases the animals exhibited very little movement for weeks during the summer (Hobbs *et al.* 2005). Dense groupings in these areas during June and July are the focus of NMFS aerial surveys, but numbers drop substantially in the upper inlet by November (Hobbs *et al.* 2005). Outside of Cook Inlet in the Gulf of Alaska beluga whale sightings are extremely rare (Laidre *et al.* 2000). Hobbs *et al.* (2005) found that tagged beluga whales moved to farther offshore during winter months, but remained within Cook Inlet. However travel distance appeared to increase during winter months, and exhibited more widely dispersed patterns both within and among individuals (Hobbs *et al.* 2005). Distribution during all months is likely influenced by prey distribution, where salmon and eulachon are concentrated in river mouths during summer months and other prey like sand lance are found in mid and bottom waters of the inlet during winter months, albeit in more dispersed patterns leading to the wider dispersal of the whales.

Based on past studies of the summer distribution of beluga whales in Cook Inlet, it appears that the population has experienced a contraction in its overall distribution (Speckman and Piatt. 2000, Hobbs *et al.* 2008, Rugh *et al.* 2010). Aerial surveys in the 1970s indicated that at least 10% of the population used areas south of Kenai River and Kalgin Island (mid- to lower Cook Inlet) during summer months, whereas more recent surveys (1993-2007) observed more than 90% of the beluga whales in upper Cook Inlet in shallow waters. According to Hobbs *et al.* (2008) 90% of the whales in the 1970s were observed within 70 nmi of the western tip of

Anchorage (Point Woronzof), whereas more recently (1998-2007) 90% were detected within 20 nmi. Although the precise reason for the range contraction is not known, the shrinking summer distribution likely reflects the reduction in the population size over the same intervals and the beluga whales' preference for dense aggregations of preferred prey species. Through the distribution has shrunk since the 1970s, possibly due to human hunting in excess of 50 individuals per year in the northern portions of Cook Inlet (where human disturbance is highest; city and port of Anchorage), surviving individuals seemed to re-occupy this habitat and abandon southern habitat (Rugh *et al.* 2010). This suggests that the northern reaches of Cook Inlet is superior habitat to southern areas, even with the anthropogenic stressors present there (Rugh *et al.* 2010). Overall, Cook Inlet beluga whale range has shrunk from greater than 7,000 km^2 to less than 3,000 km^2 (Rugh *et al.* 2010).

Habitat. During spring and summer months, beluga whales in Cook Inlet are typically concentrated near northern river mouths (Rugh *et al.* 2000). Cook Inlet experiences some of the most extreme tidal fluctuations in the world (see Hobbs *et al.* (2008) for a discussion), and beluga whales in the inlet have adapted to these tidal cycles and seemingly take advantage of them, although the precise causal reasons are not well known. Presumably, the feeding opportunities these tidal cycles proffer the beluga whale are a contributing factor. Beluga whales move further into inlets and arms during tidal floods and move back out during ebb tides in a relatively predictable manner, although highly-localized differences do exist (Ezer 2010). Tidal state is also believed to influence travel and milling behavior (Garner and Mckee. 2010). Aerial surveys and predictive models of habitat us indicate that beluga whale movement patterns are closely correlated to tidal patterns, flow accumulation and mudflats, with a preference for medium and high flow inlets of larger river basins (Goetz *et al.* 2007, Ezer *et al.* 2008). More information, however, is needed to link these habitat attributes to causative reasons for this preference. Besides feeding, studies have suggested this preference for tidal mudflats may also be attributed to calving and breeding, molting, or shelter from predators like killer whales (Calkins 1989a, Huntington 2000b, Moore *et al.* 2000c, Shelden *et al.* 2003). Individual movements suggest utilization of a variety of areas, with little to no limitation to specific areas over time (Mcguire *et al.* 2010b).

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

Calculation of beluga whale age is based on growth layers in teeth. Some debate exists as to whether a beluga whale tooth contains two growth layer groups (GLG) per year or one growth layer per year (Hobbs *et al.* 2008). Due to this ambiguity, Hobbs *et al.* (2008) summarized life history parameters according to tooth growth layers rather than years (Table 3 from (Hobbs *et al.* 2008)).

Table 3. Review of Female beluga life history parameters found in the published literature (from (Hobbs *et al.* 2008); GLG=growth layer groups).

Parameter	**Data**	**Sources**
Age at sexual maturity	8-15 GLG	(Brodie 1971, Sergeant 1973, Ognetov 1981, Seaman and Burns. 1981, Braham 1984, Burns and Seaman 1986b)
	0% at 8-9 GLGs 33% at 10-11 GLGs 94% at 12-13 GLGs	(Burns and Seaman 1986b)[a]
	9.1 +/- 2.8 GLGs	(Robeck *et al.* 2005)
Age at color change (gray to white)	12 GLGs	(Brodie 1971)
	22 GLGs	(Sergeant 1973)
Age at 1st conception	54% at 8-9 GLGs 41% at 10-11 GLGs 94% at 12-13 GLGs	(Burns and Seaman 1986b)[b]
Age at senescence	42-43 GLGs	(Brodie 1971)
Pregnancy and birth rates	Small fetuses: 0.055 at 0-11 GLGs 0.414 at 12-21 GLGs 0.363 at 22-45 GLGs 0.267 at 46-57 GLGs 0.190 at 58-77 GLGs With full-term fetuses/neonates: 0.000 at 0-11 GLGs 0.326 at 12-21 GLGs 0.333 at 22-45 GLGs 0.278 at 46-51 GLGs 0.182 at 52-57 GLGs 0.125 at 58-77 GLGs	(Burns and Seaman 1986b)
Lifespan	>60 GLGs (Oldest female estimated at 70+ GLGs)	(Burns and Seaman 1986b)
	64-65 GLGs	(Khuzin 1961, Ohsumi 1979)
	60-61 GLGs	(Brodie 1971)
	50-51 GLGs	(Sergeant 1973)
Adult annual survival	0.96-0.97	(Beland *et al.* 1992)
	0.955 (based on pilot whale data)	(Brodie 1971)
	0.935	(Lesage and Kingsley. 1998)
	0.91-0.92	(Allen and Smith. 1978)
	0.906 (includes natural & human-caused mortality)	(Burns and Seaman 1986b)
	0.84-0.905 (based on body length and lifespan	(Ohsumi 1979)
Immature annual survival	0.905 (for neonates in first half year)	(Sergeant 1973)
Reproductive rate	0.010-012	(Perrin 1982)[c]

Parameter	Data	Sources
	0.11[d]	(Burns and Seaman 1986b)
	0.13[d]	(Sergeant 1973)
	0.09[d]	(Brodie 1971)
	0.09-0.12 [d]	(Braham 1984)
	0.09-0.14[e]	(Braham 1984)
	0.12 [e]	(Sergeant 1973, Ray *et al.* 1984)
	0.08-0.14[e]	(Davis and Evans 1982)
	0.06-0.10[e]	(Davis and Finley. 1979)
	0.08-0.10[e]	(Brodie *et al.* 1981)
	0.08 (unknown)	(Breton-Provencher 1980)
Calving Interval	<3 years	(Burns and Seaman 1986b)[f]
	2 yrs and 3 years	(Sergeant 1973)[g]

[a] Alaska sample (52 whales). Sampling occring in June when most Alaskan beluga whales are born. Hobbs *et al.* (2008) note that it is possible that non-pregnant 8-9 GLGs beluga whales would have conceived before their 10-11 GLG birth date.

[b] Alaska sample of 22 whales.

[c] Based on literature review and adopted by the International Whaling Commission

[d] Based on annual calf production rates

[e] Based on calf counts

[f] For some female beluga whales. This was a tentative conclusion based on high conception rates noted in some females between the ages of 12-13 GLGs and 44-45 GLGs.

[g] Two-year intervals were for 25% of mature female belugas in eastern Canada (7 of 29 sampled); presumed after noting pregnancies occurred during lactation. Three-year intervals were for 75% of mature females in eastern Canada. Sergeant (1973) concluded that the “overlap of pregnancy and previous lactation is infrequent so that calving occurs about once in three years.”

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

Feeding. Analyses of beluga whale stomach contents indicate that beluga whales are opportunistic feeders (including including octopi, squids, crabs, shrimps, clams, mussels, snails, sandworms, and fishes such as capelin, cod, herring, smelt, flounder, sole, sculpin, lamprey, lingcod, salmon, trout, whitefish, northern pike, grayling, and tomcod (Klinkhart 1966, Fay *et al.* 1984, Haley 1986, Perez 1990, Huntington 2000a)), but specific species form the bulk of the prey when they are seasonally abundant (Hobbs *et al.* 2008). For instance, eulachon (*Thaleichthys pacificus*) also known as smelt or candlefish, are a small anadromous fish return that their natal rivers in spring for spawning. In the Susitna River, the eulachon spawning migration has a bimodal peak, with fish entering the estuary in May and again in June, and represents a significant biomass of prey, with estimates of several thousand fish entering the river in the first wave and several million entering the river in June (Calkins 1989a). The common name candlefish is derived from the fact the fish is so high in fat content during spawning, with up to 15% of total body weight as fat, that when caught and dried and strung on a wick the fish could be burned like a candle (Payne *et al.* 1999). This high fat content confers a significant source of energy for beluga whales, including calving whales that occur in the upper inlet during the same period (Calkins 1989a). The stomach contents of one beluga whale harvested in upper Cook Inlet in 1998 near the Susitna River contained only eulachon. Based on stomach sample analyses from 2002-2007 fish compose the majority of the prey species, with gadids (cod and walleye pollock) and salmonids composing the majority of the fish eaten (Hobbs *et al.* 2008). Anadromous salmonids begin concentrating at the river mouths and intertidal flats in upper Cook Inlet in late spring and early summer as emigrating smolts and immigrating adult spawners (Fried *et al.* 1979, Hazard 1988). Like eulachon, salmon are another source of lipid-rich prey for the beluga whale and represent the greatest percent frequency of occurrence of the prey species found in Cook Inlet beluga whale stomachs(Hobbs *et al.* 2008). As salmonid numbers dwindle in the fall and winter, beluga whales return to feed on nearshore or deeper water species including cod, sculpin, flounder, sole, shrimp, crab and others (Hobbs *et al.* 2008). Feeding frequently occurs collaboratively. Significantly, it appears that Cook Inlet beluga whales have switched diets during 1965 to 2007(Quakenbush and Nelson. 2010). Although the exact nature of this switch is unknown, it appears that individual whales are currently feeding at a lower trophic level than they formerly did(Quakenbush and Nelson. 2010).

Status and trends. On October 22, 2008, NMFS listed the Cook Inlet beluga whale as endangered (73 FR 62919). The Cook Inlet population is relatively isolated compared to other beluga whale stocks and the Alaska Peninsula is likely an effective physical barrier to genetic exchange, perhaps for thousands of years (Murray and Fay 1979, O'Corry-Crowe *et al.* 1997, Laidre *et al.* 2000). As such, this population may be particularly sensitive to decline. Because few robust data are available prior to the 1990s, historical population size is based upon surveys lacking strong statistical confidence. These estimates range from 150 to 450 individuals between 1963 and 1988 (Klinkhart 1966, Sergeant and Brodie 1975, Murray and Fay 1979, Calkins 1984, Hazard 1988)(Table 4). However, the most robust survey prior to 1994 occurred in 1979 and estimated Cook Inlet to contain 1,293 individuals and may have been an underestimate of the population (Calkins 1989b). Based upon this survey, the maximum stable size of the Cook Inlet population is believed to be 1,300 individuals and this value is used for management purposes.

Table 4. Estimated abundance of Cook Inlet beluga whales with coefficient of variation and 95% confidence intervals.

Year	Estimate[1]	CV	95% CI[2]	
			Lower	Upper
1979	1,293			
1994	653	0.43	291	1464
1995	491	0.44	215	1120
1996	594	0.28	347	1018
1997	440	0.14	335	578
1998	347	0.29	199	606
1999	367	0.14	279	482
2000	435	0.23	279	679
2001	386	0.087	326	458
2002	313	0.12	248	396
2003	357	0.107	290	440
2004	366	0.2	290	440
2005	278	0.18	196	394
2006	302	0.16	221	412
2007	375	0.14	285	492
2008	375	0.23	240	585
2009[2]	321	0.18	226	456

[1] All estimates, except 1979 estimate, reported in Hobbs & Shelden (2008). The 1979 estimate is from Calkins (1989a) as cited in NMFS 2008.

[2] Data from R. Hobbs, pers. comm., to A. Garrett, Apr. 2010.

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

Between 1979 and 1994, according to above noted population estimates, Cook Inlet beluga whales declined by 50%, with another 50% decline observed between 1994 and 1998. Using a growth fitted model Hobbs *et al.* (2008) observed an average annual rate of decline of -2.91% (SE = 0.010) from 1994 to 2008, and a -15.1% (SE=0.047) between 1994 and 1998. A comparison with the 1999-2008 data suggests the rate of decline at -1.45% (SE=0.014) per year (Hobbs *et al.* 2008). Reasons for population decline are somewhat nebulous, but mortality pressures include native harvests, strandings, and killer whale predation (Mahoney and Shelden 2000, Angliss and Outlaw 2006, Angliss and Outlaw 2007b, Angliss 2007). However, Cook Inlet harvests have been severely restricted (1-2 whales annually) since 1999, which coincides with a significant trend in the stabilization of the beluga whale population. Despite this, the population has not yet shown signs of recovery to historical numbers. Although not currently factors, disease and habitat disturbance have the potential to significantly impact the Cook Inlet population. According to the best available population viability analysis, there is an 80% probability that the population is declining, a 26% probability that the population will be extinct in 100 years (by 2108) and a 70% probability that the population will be extinct within 300 years (by 2308).

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

Anthropogenic threats. Cook Inlet beluga whales have been exposed to anthropogenic pressures in the past, including incidental takes in gillnet fisheries of roughly three to six individuals per year (Murray and Fay 1979, Burns and Seaman 1986a). During the early 1900s there was a short-lived commercial whaling company, The Beluga Whaling Company, which operated at the Beluga River in upper Cook Inlet. The Company during its 5 years of operation harvest 151 belugas from 1917-1921 (Mahoney and Shelden. 2000). Although evidence does not presently exist for harassment of belugas, areas of high beluga use coincide with regions of lowest ambient sound and have been known to evacuate high-use areas of humans (Huntington 2000a). Significant behavioral changes have been observed in Cook Inlet belugas in response to small boat proximity (NMFS 2006c). Propeller scars are observed on belugas, although direct mortality has not been verified. Subsistence hunting has occurred since at least the 1700s and may occur in the future (Braund and Huntington. 2010). In 1999 and 2000 there was a voluntary moratorium on subsistence harvest (Carter and Nielsen. 2010), and since substance harvest have been conducted under co-management agreements. Since 2000, no more than 2 beluga whales have been taken in subsistence harvests in any one year (Hobbs *et al.* 2008). Take levels have been as high as 20% in the 1990s, but hunts are now tightly regulated and based upon population dynamics.

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

Critical habitat. NMFS designated critical habitat for the Cook Inlet beluga whale on April 11, 2011 (76 FR 20180). Two areas specific areas are proposed comprising 7,809 square kilometers of marine habitat. Area 1 encompasses 1,918 square kilometers (741 sq. mi.) of Cook Inlet northeast of a line from the mouth of Threemile Creek (61º 08.5’N., 151 º 04.4’ W.) to Point Possession (61º 02.1’N., 150 º 24.3’ W.). This area is bounded by Anchorage, the Matansuska-Susitna Borough, and the Kenai Peninsula Borough. This area contains shallow tidal flats, river mouths or estuarine areas and is important as foraging and calving habitats. Area 1 also has the highest concentrations of beluga whales in the spring through fall as well as the greatest potential for adverse impact from anthropogenic threats. Area 1 contains many rivers with large eulachon and salmon runs, including 2 rivers in Turnagain Arm (Twenty-mile River and Placer River) which are visited by beluga whales in the early spring. Use declines in the summer and increases again in August through the fall, coinciding with coho salmon returns. Also included in Area 1 is Knik Arm and the Susitna delta. Area 2 consists of 5,891 square kilometers (2,275 sq. mi.) of Cook Inlet, located south of Area 1, north of a line at 60º 25.0’N., and includes nearshore areas south of 60º 25.0’N. along the west side of the Inlet and Kachemak Bay on the east side of the lower inlet. Area 2 is used by Cook Inlet beluga whales in a dispersed fashion for fall and winter feeding and as transit waters. Area 2 includes near and offshore areas of the mid and upper Inlet, and nearshore areas of the lower Inlet. Area 2 includes Tuxedni, Chinitna, and Kamishak Bays on the west coast and a portion of Kachemak Bay of the east coast. Dive studies indicate that beluga whales in this area dive to deeper depths and are at the surface less frequently than they are when they inhabit Area 1. The primary constituent elements essential to the conservation of

Cook Inlet beluga whales are: (1) intertidal and subtidal waters of Cook Inlet with depths <30 ft. (MLLW) and within 5 miles of high and medium flow accumulation anadromous fish streams; (2) primary prey species consisting of four species of Pacific salmon (Chinook, coho, sockeye, and chum salmon), Pacific eulachon, Pacific cod, walleye pollock, saffron cod, and yellowfin sole; (3) the absence of toxins or other agents of a type or amount harmful to beluga whales; (4) Unrestricted passage within or between the critical habitat areas; and (5) absence of in-water noise at levels result in the abandonment of habitat by Cook Inlet beluga whales. The comment period on this proposed rule closed on February 1, 2010.

Pinnipeds

Steller sea lion

Description of the species. Steller sea lions are distributed along the rim of the North Pacific Ocean from San Miguel Island (Channel Islands) off Southern California to northern Hokkaido, Japan (Loughlin *et al.* 1984, Nowak 2003). Their centers of abundance and distribution are in Gulf of Alaska and the Aleutian Islands (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; (Rice 1998a, Ban 2005, Call and Loughlin 2005). 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 (Pitcher and Calkins 1981, Calkins and Pitcher 1982b, 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 (Pike and Maxwell 1958, Gentry 1970, 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 (Merrick and Loughlin 1997, Milette 1999, Pitcher *et al.* 2001, Milette and Trites 2003, Maniscalco *et al.* 2006). 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 (Scheffer 1945, Porter 1997, Trites *et al.* 2006). Generally, female Steller sea lion will nurse their offspring until they are one to two years old (Gentry 1970, Sandegren 1970, Pitcher and Calkins 1981, Calkins and Pitcher 1982b, 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 (Pike and Maxwell 1958, Calkins and Pitcher 1982b, NMFS 1992, 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 and Pitcher 1982b, Calkins 1986, 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 2007a). 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 (Jones 1981, Pitcher and Fay 1982, Calkins and Goodwin 1988, Olesiuk *et al.* 1990b, Daniel and Schneeweis 1992, Brown *et al.* 2002, Sinclair and Zeppelin 2002, McKenzie and Wynne 2008, 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 2002).

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 US 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 US 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 and Goodwin 1988, Calkins *et al.* 1998, 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 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 (Sease *et al.* 2001, NMFS 2005d). 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 (Pascual and Adkison 1994, York 1994, Holmes and York 2003).

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, (Bonnot 1928, Rowley 1929, Scheffer 1945, Bonnot and Ripley 1948, Scheffer 1950, Pearson and Verts 1970, Bigg 1988, Atkinson *et al.* 2008, NMFS 2008g). 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 (Calkins and Goodwin 1988, Calkins *et al.* 1998, Pitcher *et al.* 1998, Trites and Donnelly 2003, Atkinson *et al.* 2008). 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 (Noda *et al.* 1995, Kim *et al.* 1996, Castellini 1999, Beckmen *et al.* 2002, NMFS 2008g). 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.

Hawaiian monk seal

Description of the species. Hawaiian monk seals are found primarily in the NWHI, which extend more than 2,000 km miles northwest of the MHI. Major breeding subpopulations occur at FFS, Laysan Island, Lisianski Island, Pearl and Hermes Reef, Midway Atoll, and Kure Atoll (Carretta *et al.* 2001). Smaller groups are found at Nihoa and Necker Islands, seals have been observed at Gardner Pinnacles, Maro Reef, and Johnston Atoll, and several dozen seals are distributed throughout MHI (Carretta *et al.* 2001, NMFS 2007b). Midway was an important breeding rookery at one time, but is no longer used (Reeves 1992). However, all Hawaiian monk seals represent a single population, with genetic connectivity high enough to maintain population-level genetic differentiation {Schultz, 2011 #79451 . Reported sightings on each of the eight MHI have become increasingly common, and births have been reported on all of the MHI except Lanai and Hawaii.

Habitat and feeding. Virtually all terrestrial substrates, including emergent reefs and shipwrecks, are used by monk seals. Sandy beaches with shallow protected water near shore are the primary haul-out areas, for pupping, nursing, and resting, although pups are born on a variety of substrates (Gilmartin 1983). Seals use vegetation behind beaches as shelter from wind and rain.

Pinniped movements are generally based on foraging. Oceanographic features, such as thermal changes that might concentrate prey densities, can affect individual seal foraging behavior (Field *et al.* 2001). Hawaiian monk seal distribution, destinations, routes, food sources, and causes of movements when not traveling between islands are not well known. Approximately 10-15% of Hawaiian monk seals migrate among the breeding populations (Johnson and Kridler 1983). Inter-island movement appears to be more likely when the islands are close together. For example, movement between Kure Atoll, Midway Atoll, Pearl and Hermes Reef appear to be fairly common, while movement between FFS and Kure Atoll (a distance of 2,000 km) is not known to occur. The western subpopulations (Pearl and Hermes Reef, Midway Islands, and Kure Atoll) exhibit a higher degree of migration compared to the more isolated subpopulations at Laysan, Lisianski, and FFS (NMFS 2007b; Table 3).

Table 5. Migration rates per Hawaiian monk seal among subpopulations per year. Data are from 1995-2008, excluding translocations.

From/To	Nihoa	Necker	FFS	Laysan	Lisianski	PHR	Midway	Kure
Nihoa	0.7727	0	0.2273	0	0	0	0	0
Necker	0	0.4259	0.5741	0	0	0	0	0
FFS	0.0027	0.0065	0.9888	0.0019	0	0	0	0
Laysan	0	0	0.0043	0.9610	0.0334	0.0013	0	0
Lisianski	0	0	0.0011	0.0484	0.9434	0.0057	0.0007	0.0007
PHR	0	0	0.0004	0.003	0.0082	0.9450	0.0305	0.0130
Midway	0	0	0	0.0013	0	0.0707	0.7639	0.1641
Kure	0	0	0	0	0.0011	0.0168	0.0786	0.9035

At the breeding islands, monk seals feed on octopus, spiny lobster, eels, and bottom-dwelling and reef fish (Gilmartin 1983; Goodman-Lowe 1998; Rice 1960). Considered foraging generalists, monk seals exhibit significant differences in diet between islands, age, and sex groups (NMFS 2007b). No research or monitoring effort has been identified that will effectively measure or index monk seal prey abundance at the major breeding atolls (NMFS 2007b).

Seals forage within the barrier reefs of the atolls and on the leeward slopes of reefs and islands, as well as nearby seamounts and submerged reefs and banks (Pullin and Stewart 2006). Several recent studies of the foraging patterns of Hawaiian monk seals near rookeries in the NWHI provide insight into their diving behavior. Dive depths appear to differ slightly between rookeries as well as between age and sex classes. Stewart *et al.* (2006) found that throughout the six NWHI breeding colonies, most dives were less than 150 meters, but found some dives exceeding 550 meters. At Pearl and Hermes Reef, most dives reach 8-40 meters, with some dives to three- to four-fold greater depths (Harington *et al.* 2004). However, at Laysan Island, this trend was reversed and dives were much deeper (800 to 1,150 feet; Harington *et al.* 2004). Most dives at FFS were to depths of 4-40 meters, but some dives exceeded 500 meters (Abernathy 1999). Parrish *et al.* (2002) noted a tendency towards night diving at FFS.

Hawaiian monk seals tend to dive within the water column, rather than to the sea floor, regardless of site (Pullin and Stewart 2006). Some work using Crittercams on seals at FFS indicates that most time spent underwater was for resting and socializing, not feeding. Despite the reef fishes of the coral shallows, adult seals forage on the slopes of the atoll and neighboring banks (Parrish *et al.* 2000). This is corroborated by the comparison between the diet composition of tagged seals and the composition of fish in each of four ecological zones (defined by depth)(Parrish and Abernathy 2006). Foraging has been shown to vary by age, with older juveniles (years 2 and 3) focusing on shallow atoll depths (10-30 meters) and yearlings feeding in sand fields at 50-100 meters. It is possible that the shift in foraging behavior with age is dependent on the physical strength to flip small rocks to find prey, rather than increasing dive duration or depth (Parrish *et al.* 2005).

Reproduction. Hawaiian monk seals do not form breeding colonies or harems (Johanos *et al.* 1994; Kenyon and Rice 1959). Mating, which occurs in water and is rarely observed, is inferred from male-female association patterns and from mounting injuries (Johanos *et al.* 1994). Breeding is asynchronous, lasting from February through September (Johanos *et al.* 1994). In recent years, fewer than 200 individuals are born annually (NMFS 2007b). Some islands, particularly in the southeastern Hawaiian Islands, have strong male-biased sex ratios, leading to female deaths from male aggression; a situation which has been significantly corrected by selective male removal (5:1, up to 25:1)(Johanos *et al.* 2010).

Females typically give birth for the first time between ages of 5 and 10 (Antonelis *et al.* 2006). Pupping patterns vary greatly and not all females give birth in consecutive years (Johanos *et al.* 1994; Kenyon and Rice 1959). Females that do give birth in consecutive years pup later each season, while females that skip a year or more give birth earlier the next season. The mean interval for births in consecutive years was found to be 381 days (Johanos *et al.* 1994). Birth rates vary depending on breeding location and year, with approximately 30-70% of all adult females giving birth in any given year (Johanos and Ragen 1994, Ragen and Lavigne 1999). Females give birth from February to August, peaking in late March/early April (Johanos *et al.* 1994), although pupping has been recorded year round. They prefer to give birth on beaches near shallow water and coral reefs surrounding the area, apparently to afford protection to the pup (Westlake and Gilmartin 1990).

Newborn pups weigh 16-17 kg and measure 95-100 cm long (Kenyon and Rice 1959). Pups are black at birth and undergo a post-natal molt late in the nursing period. Nursing lasts, on average, 39 days (Johanos *et al.* 1994), during which time the mother remains constantly near her pup in and out of water (Kenyon and Rice 1959). The mother does not eat during nursing and rapidly loses weight (Gerrodette and DeMaster 1990). After weaning, the pup is abandoned to live off of fat stores until it learns to feed on its own, while the mother swims offshore to feed (Gerrodette and Gilmartin 1990; Johnson and Johnson 1984; Kenyon and Rice 1959; Wirtz 1968). At weaning, pups normally weigh between 43-73 kg (Kenyon and Rice 1959). Rice (1964) suggested that adult females weigh approximately 205 kg and are about 2.3 m long, and the average adult male is smaller, at about 170 kg and 2.1 m.

Although nursing monk seal mothers generally avoid other adult seals, occasional pup switches do occur (Johnson and Johnson 1978, Boness 1990), and mothers sometimes foster a pup if her own is lost (Alcorn and Henderson 1984, Gerrodette *et al.* 1992). If switched pups are of similar size, survival for the first year is minimally affected; however if a larger pup switches with a small one, the larger pup will have a longer nursing period and the smaller pup’s probability of survival will be reduced (NMFS 2007b).

Status and trends. The Hawaiian monk seal was listed as endangered under the ESA on November 23, 1976 (41 FR 51611). Hawaiian monk seals are considered one of the most endangered groups of pinnipeds on the planet because all of their populations are either extinct (Caribbean monk seal) or close to extinction (Mediterranean and Hawaiian monk seals). Two periods of anthropogenic decline have been reported; the first decline occurred in the 1800s when sealers, crews of wrecked vessels, and guano and feather hunters nearly hunted monk seals to extinction (Dill and Bryan 1912, Kenyon and Rice 1959).

Following the initial collapse, expeditions to the NWHI reported increasing seal numbers and partial recovery to slightly more than 1,000 individuals (Bailey 1952, Rice 1960). However, a second decline occurred from the late 1950s to the mid-1970s; the population declined by roughly 50% by the 1980s (NMFS 1991). The total population in the FFS, Laysan Island, Lisianski Island, Pearl and Hermes Reef, Kure Atoll, and Midway, Necker, and Nihoa was estimated to be 1,501 in 1984, 1,976 seals in 1986, and 1,580 in 1992 (Ragen 1993). For the years 1985 to 1993 the mean beach counts declined by approximately 5% per year. This downward trend is expected to continue, mainly because of poor pup and juvenile survival in recent years.

The best estimate of the total population of the species is 1,202 seals and the minimum population size estimate for the Hawaiian monk seal is 1,176 seals (NMFS 2007b). Data collected in 2008 suggest that the species population is now 1,146 (NMFS 2009a). A log-linear regression of estimated abundance from 1998 to 2006 suggests the population has declined on average -3.9% per year, and models predict that the total population of the species will fall below 1,000 monk seals within 5 years (NMFS 2007b). Trends in abundance vary considerably among the six main subpopulations.

A recent five-year status review conducted by NMFS recommends that the Hawaiian monk seals' endangered status should remain the same (72 FR 46966, August 2007). The population dynamics at the different subpopulations have varied considerably, and current demographic variability among the island populations probably reflects a combination of different histories of human disturbance and management (Gerrodette and DeMaster 1990, Craig and Ragen 1999), and varying environmental conditions (Baker *et al.* 2007; Baker and Thompson 2007; Craig and Ragen 1999; Polovina *et al.* 1994). The current status of the Hawaiian monk seal is dire, due to low juvenile survival and the number of aging breeding females in the population. Consequently, NMFS is currently exploring development of a captive care program for juvenile Hawaiian monk seals to enhance their potential for survival and recovery (NMFS 2007b).

The total of mean, non-pup, beach counts at the main reproductive subpopulations in 2005 was approximately 67% lower than in 1958 (Benson *et al.* 2007b). A log-linear regression of estimated abundance from 1998 (the first year for which a reliable total abundance estimate was obtained) to 2006 estimates that abundance declined by 3.9% annually (Figure 3)(NMFS 2007b).

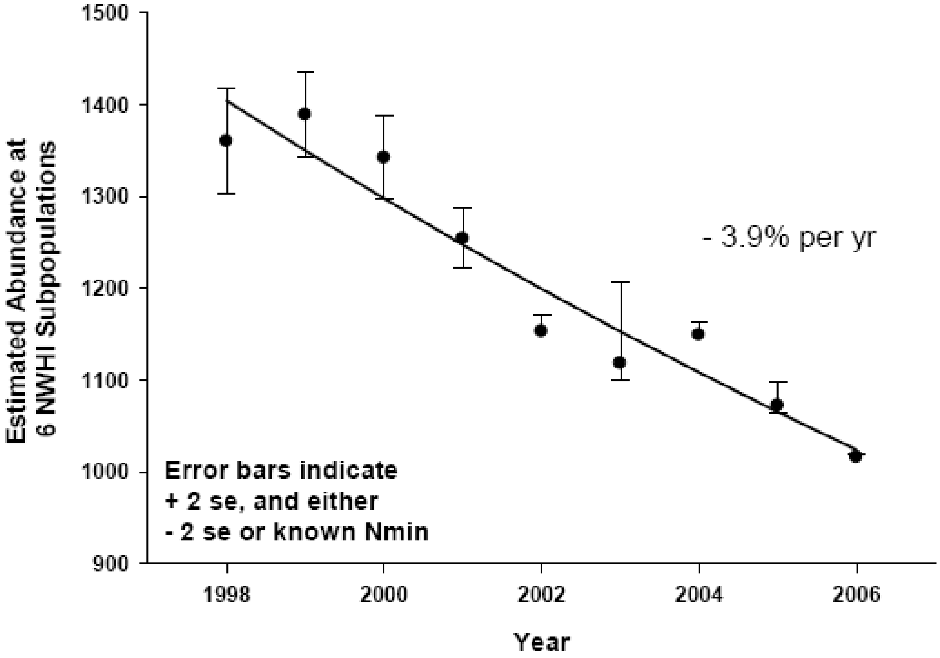

Figure 3. Trends in abundance of Hawaiian monk seals at the six main NWHI subpopulations combined, 1998-2006. This graph does not include abundance estimates for Necker, Nihoa, or the MHI. Error bars indicate ±2 standard errors or known minimum abundance. The fitted trend line reveals an estimated decline of 3.9% (NMFS 2007b).

Trends vary among the six main subpopulations. Non-pup beach counts at FFS have decreased by 73% from 1989 to 2005 (Benson *et al.* 2007b). At one time, FFS accounted for over 50% of the total non-pup beach counts among the NWHI subpopulations; however, that proportion has dropped to 25%, although FFS still maintains the single largest subpopulation (NMFS 2007b). The annual number of births has dropped from a high of 127 in 1988 to 39 in 2006, and survival from weaning to age two has declined from a high near 90% in the mid-1980s to a low of 8% in 1997 (NMFS 2007b). Shark predation and prey availability are two potentially responsible factors.

Populations at Laysan and Lisianski Islands declined sharply after the late 1950s. In 1994, 21 adult male Hawaiian monk seals were relocated from Laysan Island to the MHI in an attempt to equalize the sex ratio at Laysan Island, and beach counts increased from 1995 to 2000, but have declined in the following years, while the Lisianski subpopulation has remained relatively stable, yet low, since the 1970s. Marine debris and low fecundity are factors that might contribute to the lack of subpopulation growth at Lisianski Island. And while the decline in abundance in Laysan may be related to female mortality caused by male aggression, juvenile survival is relatively good for most cohorts, and the lack of recovery on Laysan is not understood (NMFS 2007b).

Until recently, the three westernmost subpopulations, Kure, Midway and Pearl and Hermes Reef exhibited substantial growth. Beach counts on Kure increased 5% per year from 1983 to 2000, declined in 2000-2001, and are now slowly increasing. At Midway, beach counts increased from 1995 to 2000, and have since declined. The subpopulation at Pearl and Hermes Reef increased after the mid-1970s. Prior to 1999, beach count increases of up to 7% per year were observed. This is the highest estimate of the maximum net productivity rate observed for this species. Since 2000, low juvenile survival, thought to be due largely to food limitation, has been widespread with rare exceptions in the NWHI, resulting in the population decline (Benson *et al.* 2007b), and several recent cohorts at the three westernmost sites indicate a drop in juvenile survival (NMFS 2007b).

Sightings of Hawaiian monk seals have occurred on at least three occasions at the remote Pacific location of Johnston Atoll (excluding nine adult males relocated there from Laysan Island in 1984).

The decrease in survival rates of immature animals, including a decline in survival from birth to weaning, and survival from weaning to age 2 years has contributed to a dramatically imbalanced age structure for all six of the main NWHI subpopulations (Figure 4)(NMFS 2007b, 2009a). Although studies show that the relationship between size of pups and first year survival vary between subpopulations and over time, site-specific analyses do support girth and year as predictors of first-year survival at each location. When conditions for survival are worse, the relationship between size and survival strengthens. The simplest explanation for this is food limitation (Baker 2008).

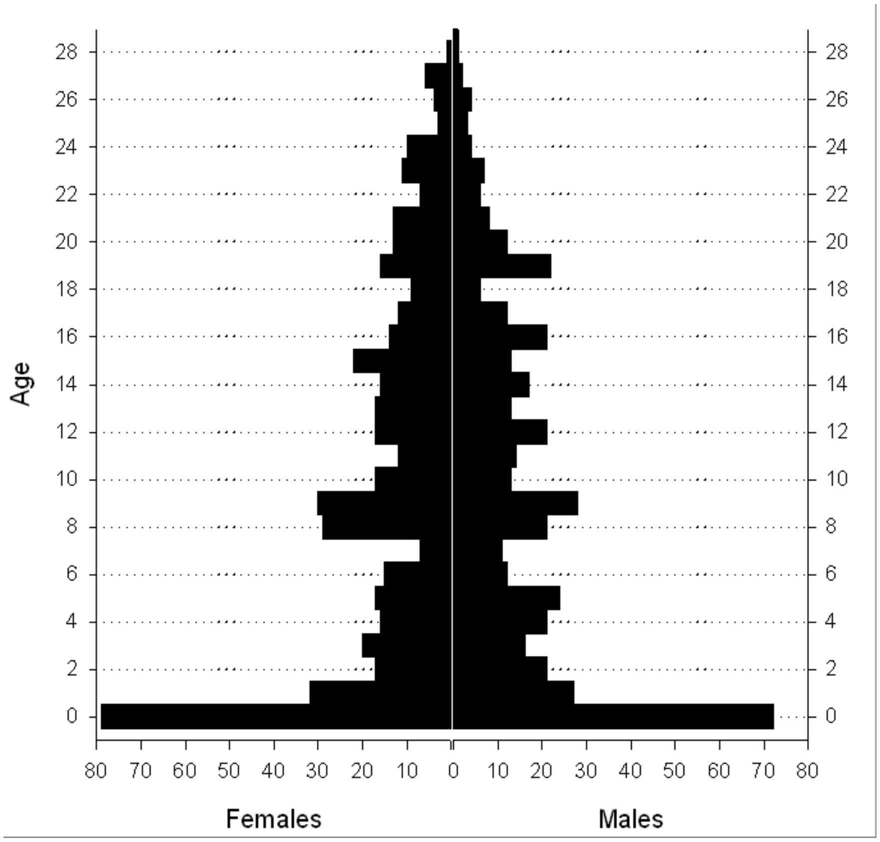

Figure 4. Age distribution for the Hawaiian monk seal population in the NWHI (MMRP unpublished data in NMFS 2009a).

Sightings and births are increasing in the MHI, although systematic surveys were not conducted before 2000, and counts do not represent total abundance, as they do not account for seals in the water, and not every seal on land is detected. In 2000, the count in the MHI was 45 seals, and in 2001, 52 were counted. In 2005, the total number of unique seals identified was 77, based on non-systematic sightings. Annual births have increased since the mid-1990s. Although this could be a positive indication for the survival of the species, the increased chance of contraction of diseases such as leptospirosis and toxoplasmosis from wild and domestic animals, and increased interactions with humans, including fishermen, boaters, and divers raise conservation concerns which do not apply to the NWHI (NMFS 2007b). The only available estimate of abundance in the MHI is 152 individuals, with an annual population growth rate of 7% {Baker, 2011 #79839}. Survival to one year of age is 77% in the NHI, versus 42-57% in the NWHI (Baker *et al.* 2011).

Hawaiian monk seal pups weaned in the MHI exhibit higher girths and lengths compared to pups from the NWHI, as a result of pre- and post-partum maternal investment – a partial reflection of prey availability – contradicting the studies that indicate better foraging conditions in the NWHI. Suggested explanations for this include a higher per capita availability of prey in the MHI, similar absolute preferred prey densities when apex predators are not included in the biomass, and increased prey availability due to reduced competition from apex predators (Baker and Johanos 2004).

Critical habitat. Critical habitat was originally designated on April 30, 1986 (51 FR 16047), and was extended on May 26, 1988 (53 FR 18988; CFR 226.201). The critical habitat includes all beach areas, sand spits and islets, including all beach crest vegetation to its deepest extent inland, lagoon waters, inner reef waters, and ocean waters out to a depth of 20 fathoms (37 m) around the following: Kure Atoll (28°24′ N, 178°20′ W), Midway Islands, except Sand Island and its harbor (28°14′ N, 177°22′ W), Pearl and Hermes Reef (27°55′ N, 175° W), Lisianski Island (26°46′ N, 173°58′ W), Laysan Island (25°46′ N, 171°44′ W), Maro Reef (25°25′ N, 170°35′ W), Gardner Pinnacles (25°00′ N, 168°00′ W), French Frigate Shoals (23°45′ N, 166°00′ W), Necker Island (23°34′ N, 164°42′ W), Nihoa Island (23°03.5′ N, 161°55.5′ W). 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. Both components are currently under significant degradation pressure. Because the marine critical habitat is in the Papahānaumokuākea Marine National Monument, fishing is forbidden in the critical habitat. A result of this is the establishment of large predatory fishes (sharks and jacks) that compete with Hawaiian monk seals for prey resources. This may be a cause of seal malnourishment seen throughout many islands in the region.

On June 2, 2011, NMFS proposed to extend critical habitat in the NWHI to Sand Island (Midway) and ocean waters out to a depth of 500 (76 FR 27988). The area around the MHI (Kaula Island, Niihau, Kauai, Oahu, Maui Nui, and Hawaii) to a deth of 500 m and inland to a distance of 5 m from shore was also proposed.

Sea turtles

Leatherback sea turtle

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

Leatherbacks also occur in Mediterranean and Indian Ocean waters (Casale 2003, Casale *et al.* 2003, Hamann *et al.* 2006a). Associations exist with continental shelf and pelagic environments and sightings occur in offshore waters of 7-27˚ C (CETAP 1982). Juvenile leatherbacks usually stay in warmer, tropical waters >21˚ C (Eckert 2002). Males and females show some degree of natal homing to annual breeding sites (James *et al.* 2005).

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

Atlantic Ocean. Previous genetic analyses of leatherbacks using only mitochondrial DNA (mtDNA) resulted in an earlier determination that within the Atlantic basin there are at least three genetically different nesting populations: the St. Croix nesting population (US Virgin

Islands), the mainland nesting Caribbean population (Florida, Costa Rica, Suriname/French Guiana), and the Trinidad nesting population (Dutton *et al.* 1999). Further genetic analyses using microsatellite markers in nuclear DNA along with the mtDNA data and tagging data has resulted in Atlantic Ocean leatherbacks now being divided into seven groups or breeding populations: Florida, Northern Caribbean, Western Caribbean, Southern Caribbean/Guianas, West Africa, South Africa, and Brazil (TEWG 2007a).

Caribbean Sea. Nesting occurs in Puerto Rico, St. Croix, Costa Rica, Panama, Colombia, Trinidad and Tobago, Guyana, Suriname, and French Guiana (Márquez 1990, Spotila *et al.* 1996, Bräutigam and Eckert 2006b).

Indian Ocean. Nesting is reported in South Africa, India, Sri Lanka, and the Andaman and Nicobar islands(Hamann *et al.* 2006a).

Pacific Ocean. Leatherbacks are found from tropical waters north to Alaska within the North Pacific and is the most common sea turtle in the eastern Pacific north of Mexico (Stinson 1984a, Eckert 1993a, Wing and Hodge 2002). The west coast of Central America and Mexico hosts nesting from September-March, although Costa Rican nesting peaks during April-May (Chacón-Chaverri and Eckert 2007, LGL Ltd. 2007). Leatherback sea turtles disappeared from India before 1930, have been virtually extinct in Sri Lanka since 1994, and appear to be approaching extinction in Malaysia (Spotila *et al.* 2000). In Fiji, Thailand, and Australia, leatherback sea turtles have only been known to nest in low densities and scattered sites. Leatherback nesting aggregations occur widely in the Pacific, including China, Malaysia, Papua New Guinea, Indonesia, Thailand, Australia, Fiji, the Solomon Islands, and Central America (Limpus 2002, Dutton *et al.* 2007). Significant nesting also occurs along the Central American coast (Márquez 1990). Although not generally known to nest on Japanese shores, two nests were identified in the central Ryukyu Islands in 2002 (Kamezaki *et al.* 2002).

Nesting beaches also occur in Mexico and Costa Rica (nesting occurs October through March) are a separate population from the western Pacific beaches. In Costa Rica, leatherbacks nest at Playa Naranjo in Santa Rosa National Park, the second-most important nesting beach on the Pacific coast (Yañez *et al.* 2010), Rio Oro on the Osa Peninsula, and at various beaches in Las Baulas National Park, which includes Playa Langosta and Playa Grande and contains the largest colony of leatherbacks in the Pacific (Spotila 2004a). Females typically lay six clutches per season (average nine days between nests), which incubate for 58–65 days (Lux *et al.* 2003). Limited nesting also occurs along Nicaragua, Panama, El Salvador, Vanuatu, and Guatemala.

Growth and reproduction. It has been thought that leatherbacks reach sexual maturity somewhat faster than other sea turtles (except Kemp's ridley), with an estimated range of 3-6 years (Rhodin 1985) to 13-14 years (Zug and Parham 1996). However, recent research suggests otherwise, with western North Atlantic leatherbacks possibly not maturing until as late as 29 years of age (Avens and Goshe 2007). Female leatherbacks nest frequently (up to 10 nests per year and about every 2-3 years). During each nesting, females produce 100 eggs or more per clutch and 700 eggs or more per nesting season (Schultz 1975). However, up to ~30% of the eggs can be infertile. Thus, the actual proportion of eggs that can result in hatchlings is less than this seasonal estimate. The eggs incubate for 55-75 days before hatching.

Habitat. Leatherbacks occur throughout marine waters, from nearshore habitats to oceanic environments (Schroeder and Thompson 1987, Shoop and Kenney 1992a, Grant and Ferrell 1993, Starbird *et al.* 1993). Movements are largely dependent upon reproductive and feeding cycles and the oceanographic features that concentrate prey, such as frontal systems, eddy features, current boundaries, and coastal retention areas (Collard 1990, Davenport and Balazs 1991, Frazier 2001, HDLNR 2002, Benson *et al.* 2011). Aerial surveys off the western US support continental slope waters as having greater leatherback occurrence than shelf waters (Green *et al.* 1992, Carretta and Forney 1993, Green *et al.* 1993, Bowlby *et al.* 1994). Nesting sites appear to be related to beaches with relatively high exposure to wind or wind-generated waves (Santana Garcon *et al.* 2010).

Areas above 30º N in the Atlantic appear to be popular foraging locations (Fossette *et al.* 2009b). Northern foraging areas were proposed for waters between 35º and 50º N along North American, Nova Scotia, the Gulf of Saint-Laurent, in the western and northern Gulf Stream, the Northeast Atlantic, the Azores front and northeast of the Azores Islands, north of the Canary Islands. Southern foraging was proposed to occur between 5º and 15º N in the Mauritania upwelling, south of the Cape Verde islands, over the Guinea Dome area, and off Venezuela, Guyana and Suriname.

Migration and movement. Leatherback sea turtles migrate throughout open ocean convergence zones and upwelling areas, along continental margins, and in archipelagic waters (Morreale *et al.* 1994, Eckert 1998, 1999). In a single year, a leatherback may swim more than 9,600 km to nesting and foraging areas throughout ocean basins (Eckert 1998, Ferraroli *et al.* 2004, Hays *et al.* 2004, Eckert 2006, Eckert *et al.* 2006, Sale *et al.* 2006, Benson *et al.* 2007a, Benson *et al.* 2007c). Much of this travel may be due to movements within current and eddy features, moving individuals along (Sale and Luschi 2009). Return to nesting beaches may be accomplished by a form of geomagnetic navigation and use of local cues (Sale and Luschi 2009). Leatherback females will either remain in nearshore waters between nesting events, or range widely, presumably to feed on available prey (Byrne *et al.* 2009, Fossette *et al.* 2009a).
Fossette *et al.* (2009b) identified three main migratory strategies in leatherbacks in the North Atlantic (almost all of studied individuals were female). One involved 12 individuals traveling to northern latitudes during summer/fall and returning to waters during winter and spring. Another strategy used by six individuals was similar to this, but instead of a southward movement in fall, individuals overwintered in northern latitudes (30-40º N, 25-30º W) and moved into the Irish Sea or Bay of Biscay during spring before moving south to between 5 and 10º in winter, where they remained or returned to the northwest Atlantic. A third strategy, which was followed by three females remaining in tropical waters for the first year subsequent to nesting and moving to northern latitudes during summer/fall and spending winter and spring in latitudes of 40-50º N.

Satellite tracking data reveal that leatherback females leaving Mexican and Central American nesting beaches migrate towards the equator and into Southern Hemisphere waters, some passing the Galápagos Islands, and disperse south of 10ºS (Dutton *et al.* 2006, Shillinger *et al.* 2010). However, observations of leatherbacks in the Galápagos Islands are rare (Zárate *et al.* 2010). Nesting site selection in the southwest Pacific appears to favor sites with higher wind and wave exposure, possibly as a means to aid hatchling dispersal (Garcon *et al.* 2010). Individuals nesting in Malayasia undergo migrations to tropical feeding areas, taking 5-7 months to arrive there from nesting locations (Benson *et al.* 2011). Additional foraging occurs in temperate locations, including across the Pacific basin aloing the US west coast; individuals take 10-12 months to migrate here (Benson *et al.* 2011). Individuals nesting during the boreal summer move to feeding areas in the North China Sea, while boreal winter nesters moved across the Equator to forage in the Southern Hemisphere (Benson *et al.* 2011).

Sex ratio. A significant female bias exists in all leatherback populations thus far studied. An examination of strandings and in-water sighting data from the US Atlantic and Gulf of Mexico coasts indicates that 60% of individuals were female. Studies of Suriname nesting beach temperatures suggest a female bias in hatchlings, with estimated percentages of females hatched over the course of each season at 75.4, 65.8, and 92.2% in 1985, 1986, and 1987, respectively (Plotkin 1995). Binckley *et al.* (1998) found a heavy female bias upon examining hatchling gonad histology on the Pacific coast of Costa Rica, and estimated male to female ratios over three seasons of 0:100, 6.5:93.5, and 25.7:74.3. James *et al.* (2007) also found a heavy female bias (1.86:1) as well as a primarily large sub-adult and adult size distribution. Leatherback sex determination is affected by nest temperature, with higher temperatures producing a greater proportion of females (Mrosovksy 1994, Witzell et al. 2005b).

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

Status and trends. Leatherback sea turtles received protection on June 2, 1970 (35 FR 8491) under the Endangered Species Conservation Act and, since 1973, have been listed as endangered under the ESA, but declines in nesting have continued worldwide. Breeding females were initially estimated at 29,000-40,000, but were later refined to ~115,000 (Pritchard 1971, 1982). Spotila *et al.* (1996) estimated 34,500 females, but later issued an update of 35,860 (Spotila 2004b). The species as a whole is declining and local populations are in danger of extinction (NMFS 2001b, a).

Nesting aggregations occur along Gabon, Sao Tome and Principe, French Guiana, Suriname, and Florida (Márquez 1990, Spotila *et al.* 1996, Bräutigam and Eckert 2006b). Widely dispersed but fairly regular African nesting also occurs between Mauritania and Angola (Fretey *et al.* 2007). Many sizeable populations (perhaps up to 20,000 females annually) of leatherbacks are known to nest in West Africa (Fretey 2001b). The population of leatherbacks nesting on Gabon beaches has been suggested as being the world's largest, with 36,185-126,480 clutches being laid by 5,865-20,499 females annually from 2002-2007 (Witt *et al.* 2009). The total number of females utilizing Gabon nesting beaches is estimated to be 15,730- 41,373 (Witt *et al.* 2009). North Atlantic leatherbacks likely number 34,000-94,000 individuals, with females numbering 18,800 and the eastern Atlantic segment numbering 4,700 (TEWG 2007a). Trends and numbers include only nesting females and are not a complete demographic or geographic cross-section. In 1996, the entire Western Atlantic population was characterized as stable at best (Spotila *et al.* 1996), with roughly 18,800 nesting females. A subsequent analysis indicated that by 2000, the western Atlantic nesting population had decreased to about 15,000 nesting females (NMFS 2011l). Spotila *et al.* (1996) estimated that the entire Atlantic basin, including all nesting beaches in the Americas, the Caribbean, and West Africa, totaled approximately 27,600 nesting females, with an estimated range of 20,082-35,133. This is consistent with other estimates of 34,000-95,000 total adults (20,000-56,000 adult females; 10,000-21,000 nesting females)(TEWG 2007b).

The largest nesting aggregation in the western North Atlantic occurs in French Guiana and Suriname, likely belongs to a metapopulation whose limits remain unknown (Rivalan *et al.* 2006). Heppell *et al.* (2003a) concluded that leatherbacks generally show less genetic structuring than green and hawksbill sea turtles. The French Guiana nesting aggregation has declined ~15% annually since 1987 (NMFS 2001a). However, from 1979-1986, the number of nests increased ~15% annually, possibly indicating the current decline may be linked with the erosion cycle of Guiana beaches (NMFS 2006e). Guiana nesting may have increased again in the early 2000s (NMFS 2006e). Suriname nesting numbers have recently increased from more than 10,000 nests annually since 1999 and a peak of 30,000 nests in 2001. Overall, Suriname and French Guiana nesting trends towards an increase (Hilterman and Goverse 2003, Girondot *et al.* 2007). Florida (March-July) and US Caribbean nesting since the early 1980s has increased ~0.3% and 7.5% per year, respectively, but lags behind the French Guiana coast and elsewhere in magnitude (NMFS/SEFSC 2001). This positive growth was seen within major nesting areas for the stock, including Trinidad, Guyana, and the combined beaches of Suriname and French Guiana (TEWG 2007b). Using both Bayesian modeling and regression analyses, the TEWG (2007b) determined that the Southern Caribbean/Guianas stock had demonstrated a long-term, positive population growth rate (using nesting females as a proxy for population).

The Caribbean coast of Costa Rica and extending through Chiriquí Beach, Panama, represents the fourth largest known leatherback rookery in the world (Troeng *et al.* 2004). Examination of data from three index nesting beaches in the region (Tortuguero, Gandoca, and Pacuare in Costa Rica) using various Bayesian and regression analyses indicated that the nesting population likely was not growing during 1995-2005 (TEWG 2007b). Other modeling of the nesting data for Tortuguero indicates a 67.8% decline between 1995 and 2006 (Troëng *et al.* 2007).

In Puerto Rico, the primary nesting beaches are at Fajardo and on the island of Culebra. Nesting between 1978 and 2005 ranged between 469-882 nests, and the population has been growing since 1978, with an overall annual growth rate of 1.1% (TEWG 2007b). At the primary nesting beach on St. Croix, the Sandy Point National Wildlife Refuge, nesting has fluctuated from a few hundred nests to a high of 1,008 in 2001, and the average annual growth rate has been approximately 1.1% from 1986-2004 (TEWG 2007b).

The Florida nesting stock comes ashore primarily along the east coast of Florida. This stock is of growing importance, with total nests between 800-900 per year in the 2000s following nesting totals fewer than 100 nests per year in the 1980s (NMFS 2011l). Using data from the index nesting beach surveys, the TEWG (2007b) estimated a significant annual nesting growth rate of 1% between 1989 and 2005. Stewart *et al.* (2011) evaluated nest counts from 68 Florida beaches over 30 years (1979-2008) and found that nesting increased at all beaches with trends ranging from 3.1%-16.3% per year, with an overall increase of 10.2% per year. In 2007, a record 517 leatherback nests were observed on the index beaches in Florida, with 265 in 2008, and then an increase to a new record of 615 nests in 2009, and a slight decline in 2010 back to 552 nests (FWC Index Nesting Beach database). This up-and-down pattern is thought to be a result of the cyclical nature of leatherback nesting, similar to the biennial cycle of green turtle nesting.

The most recent population estimate for leatherback sea turtles from the North Atlantic as a whole is between 34,000-90,000 adult individuals (20,000-56,000 adult females)(TEWG 2007b).

Heavy declines have occurred at all major Pacific basin rookeries, as well as Mexico, Costa Rica, Malaysia, India, Sri Lanka, Thailand, Trinidad, Tobago, and Papua New Guinea. This includes a nesting decline of 23% between 1984-1996 at Mexiquillo, Michoacán, Mexico (Sarti *et al.* 1996). According to reports from the late 1970s and early 1980s, three beaches on the Pacific coast of Mexico supported as many as half of all leatherback turtle nests for the eastern Pacific. Since the early 1980s, the eastern Pacific Mexican population of adult female leatherback turtles has declined to slightly more than 200 individuals during 1998-1999 and 1999-2000 (Sarti *et al.* 2000). Spotila *et al.* (2000) reported the decline of the leatherback turtle population at Playa Grande, Costa Rica, which had been the fourth largest nesting colony in the world. Between 1988 and 1999, the nesting colony declined from 1,367 to 117 female leatherback turtles. Based on their models, Spotila *et al.* (2000)estimated that the colony could fall to less than 50 females by 2003-2004. Fewer than 1,000 females nested on the Pacific coast of Mexico from 1995-1996 and fewer than 700 females are estimated for Central America (Spotila *et al.* 2000). The number of leatherback turtles nesting in Las Baulas National Park declined rapidly during the 1990s, from about 1,500 females during the 1988–89 nesting season, to about 800 in 1990–91 and 1991–92 to 193 in 1993–94 (Williams *et al.* 1996) and 117 in 1998–99 (Spotila *et al.* 2000). Spotila (2004a) reported that between 59 and 435 leatherbacks nest at Las Baulas each year depending on the El Niño–La Niña cycle. Only an Indonesian nesting assemblage has remained relatively abundant in the Pacific basin. The largest extant leatherback nesting assemblage in the Indo-Pacific lies on the northern Vogelkop coast of Irian Jaya (West Papua), Indonesia, with roughly 3,000 nests recorded annually (Putrawidjaja 2000, Suárez *et al.* 2000) (Dutton *et al.* 2007). The Western Pacific leatherback metapopulation harbors the last remaining nesting aggregation of significant size in the Pacific with approximately 2700–4500 breeding females (Dutton *et al.* 2007, Hitipeuw *et al.* 2007). The total number of nests per year for the Jamursba-Medi leatherback nesting population ranged between a high of 6,373 nests in 1996 and a low of 1,537 nests in 2010 (Hitipeuw *et al.* 2007).

Declines in the western Pacific is equally severe. Nesting at Terengganu, Malaysia is 1% of that in 1950s (Chan and Liew 1996). The South China Sea and East Pacific nesting colonies have undergone catastrophic collapse. Overall, Pacific populations have declined from an estimated

81,000 individuals to <3,000 total adults and subadults (Spotila *et al.* 2000). The number of nesting leatherbacks has declined by an estimated 95% over the past 20 years in the Pacific (Gilman 2009). Drastic overharvesting of eggs and mortality from fishing activities is likely responsible for this tremendous decline (Sarti *et al.* 1996, Eckert 1997).

Based on the survey and tagging work, it was estimated that 400-500 female leatherbacks nest annually on Great Nicobar Island (Andrews *et al.* 2002). The number of nesting females using the Andaman and Nicobar Islands combined was estimated around 1,000 (Andrews and Shanker 2002).

Natural threats. Sea turtles face predation primarily by sharks and to a lesser extent by killer whales (Pitman and Dutton 2004). Hatchlings are preyed upon by herons, gulls, dogfish, and sharks. Leatherback hatching success is particularly sensitive to nesting site selection, as nests that are overwashed have significantly lower hatching success and leatherbacks nest closer to the high-tide line than other sea turtle species (Caut *et al.* 2009b).

Anthropogenic threats. Leatherback nesting and marine environments are facing increasing impacts through widespread development and tourism along nesting beaches (Hamann *et al.* 2006a, Maison 2006, Hernandez *et al.* 2007, Santidrián Tomillo *et al.* 2007). Structural impacts to beaches include building and piling construction, beach armoring and renourishment, and sand extraction (Lutcavage *et al.* 1997b, Bouchard *et al.* 1998). In some areas, timber and marine debris accumulation as well as sand mining reduce available nesting habitat (Chacón Chaverri 1999, Formia *et al.* 2003, Laurance *et al.* 2008, Bourgeois *et al.* 2009). Lights on or adjacent to nesting beaches alter nesting adult behavior and is often fatal to emerging hatchlings as they are drawn to light sources and away from the sea (Witherington and Bjorndal 1991, Witherington 1992, Cowan *et al.* 2002, Deem *et al.* 2007, Bourgeois *et al.* 2009). Plastic ingestion is very common in leatherbacks and can block gastrointestinal tracts leading to death (Mrosovsky *et al.* 2009). Along the coast of Peru, 13% of 140 leatherback carcasses were found to contain plastic bags and film (Fritts 1982). Although global warming may expand foraging habitats into higher latitude waters, increasing temperatures may increase feminization of nests (Mrosovsky *et al.* 1984, James *et al.* 2006, McMahon and Hays 2006, Hawkes *et al.* 2007b). Rising sea levels may also inundate nests on some beaches. Egg collection is widespread and attributed to catastrophic declines, such as in Malaysia. Harvest of females along nesting beaches is of concern worldwide.

Bycatch, particularly by longline fisheries, is a major source of mortality for leatherback sea turtles (Crognale *et al.* 2008, Gless *et al.* 2008, Fossette *et al.* 2009a, Petersen *et al.* 2009). Wallace *et al.* (2010) estimated that between 1990 and 2008, at least 85,000 sea turtles were captured as bycatch in fisheries worldwide. This estimate is likely at least two orders of magnitude low, resulting in a likely bycatch of nearly half a million sea turtles annually (Wallace *et al.* 2010); many of these turtles are expected to be leatherbacks. Donoso and Dutton (2010) found that 284 leatherbacks were bycaught between 2001 and 2005 as part of the Chilean longline fishery, with two individuals observed dead; leatherbacks were the most frequently bycaught sea turtle species. Between 8-17 leatherback turtles likely died annually between 1990 and 2000 in interactions with the California/Oregon drift gillnet fishery; 500 leatherback turtles are estimated to die annually in Chilean and Peruvian fisheries; 200 leatherback turtles are estimated to die in direct harvests in Indonesia; and, before 1992, the North Pacific driftnet fisheries for squid, tuna, and billfish captured an estimated 1,000 leatherback turtles each year, killing about 111 of them each year . An estimated 6,363 leatherback sea turtles were documented as caught by the US Atlantic tuna and swordfish longline fisheries between 1992-1999 (NMFS SEFSC 2001). Currently, the US tuna and swordfish longline fisheries managed under the HMS FMP are estimated to capture 1,764 leatherbacks (no more than 252 mortalities) for each 3-year period starting in 2007 (NMFS 2004). In 2010, there were 26 observed interactions between leatherback sea turtles and longline gear used in the HMS fishery (Garrison and Stokes 2011a, 2011b). All leatherbacks were released alive, with all gear removed for the majority of captures. While 2010 total estimates are not yet available, in 2009, 285.8 (95% CI: 209.6-389.7) leatherback sea turtles are estimated to have been taken in the longline fisheries managed under the HMS FMP based on the observed takes (Garrison and Stokes 2010). Lewison *et al.* (2004) estimated that 30,000-60,000 leatherbacks were taken in all Atlantic longline fisheries in 2000 (including the US Atlantic tuna and swordfish longline fisheries, as well as others). Finkbeiner *et al.* (2011) estimated that annual bycatch interactions total 1,400 individuals annually for US Atlantic fisheries (resulting in roughly fourty mortalities) and one hundred interactions in US Pacific fisheries (resulting in about ten mortalities). Mortality of leatherbacks in the US shrimp fishery is now estimated at 54 turtles per year. Data collected by the NEFSC Fisheries Observer Program from 1994 through 1998 (excluding 1997) indicate that a total of 37 leatherbacks were incidentally captured (16 lethally) in drift gillnets set in offshore waters from Maine to Florida during this period. Observer coverage for this period ranged from 54 to 92%. Trinidad and Tobago's Institute for Marine Affairs estimated that more than 3,000 leatherbacks were captured incidental to gillnet fishing in the coastal waters of Trinidad in 2000. Half or more of the gravid turtles in Trinidad and Tobago waters may be killed (Lee Lum 2003), though many of the turtles do not die as a result of drowning, but rather because the fishermen butcher them in order to get them out of their nets (NMFS 2001b).

We know little about the effects of contaminants on leatherback sea turtles. The metals arsenic, cadmium, copper, mercury, selenium, and zinc bioaccumulate, with cadmium in highest concentration in leatherbacks versus any other marine vertebrate (Gordon *et al.* 1998, Caurant *et al.* 1999). A diet of primarily jellyfish, which have high cadmium concentrations, is likely the cause (Caurant *et al.* 1999). Organochlorine pesticides have also been found (Mckenzie *et al.* 1999). PCB concentrations are reportedly equivalent to those in some marine mammals, with liver and adipose levels of at least one congener being exceptionally high (PCB 209: 500-530 ng/g wet weight Davenport *et al.* 1990, Oros *et al.* 2009).

Critical habitat. On March 23, 1979, leatherback critical habitat was identified adjacent to Sandy Point, St. Croix, US Virgin Islands from the 183 m isobath to mean high tide level between 17° 42’12” N and 65°50’00” W (44 FR 17710). This habitat is essential for nesting, which has been increasingly threatened since 1979, when tourism increased significantly, bringing nesting habitat and people into close and frequent proximity. However, studies do not currently support significant critical habitat deterioration.

On Janauary 5, 2010, the NMFS proposed to designate critical habitat for leatherback sea turtles in waters along Washington State (Cape Flattery to the Umpqua River; 63,455 km^2) and California (Point Arena to point Vincente; 119,400 km^2). The primary constituent elements of these areas 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.

Hawksbill sea turtle

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

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

Migration and movement. Upon first entering the sea, neonatal hawksbills in the Caribbean are believed to enter an oceanic phase that may involve long distance travel and eventual recruitment to nearshore foraging habitat (Boulon 1994). In the marine environment, the oceanic phase of juveniles (i.e., the "lost years") remains one of the most poorly understood aspects of hawksbill life history, both in terms of where turtles occur and how long they remain oceanic. Nesting site selection in the southwest Pacific appears to favor sites with higher wind and wave exposure, possibly as a means to aid hatchling dispersal(Garcon *et al.* 2010).

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

Hawksbill sea turtles appear to be rare visitors to the Gulf of Mexico, with Florida being the only Gulf state with regular sightings (Rabalais and Rabalais 1980, Hildebrand 1983, Witzell 1983, NMFS and USFWS 1993, Rester and Condrey 1996). Individuals stranded in Texas are generally young (hatchlings or yearlings) originating from Mexican nesting beaches (Hildebrand 1983, Amos 1989, Collard and Ogren 1990, Landry and Costa 1999).

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

The life history of hawksbills consists of a pelagic stage that lasts from hatching until they are approximately 22-25 cm in straight carapace length (Meylan 1988, Meylan and Donnelly 1999), followed by residency in coastal developmental habitats.

Feeding. Dietary data from oceanic stage hawksbills are limited, but indicate a combination of plant and animal material (Bjorndal 1997b).

Status and trends. Hawksbill sea turtles received protection on June 2, 1970 (35 FR 8495) under the Endangered Species Conservation Act and since 1973 have been listed as endangered under the ESA. Although no historical records of abundance are known, hawksbill sea turtles are considered to be severely depleted due to the fragmentation and low use of current nesting beaches (NMFS and USFWS 2007e). Worldwide, an estimated 21,212-28,138 hawksbills nest each year among 83 sites. Among the 58 sites for with historic trends, all show a decline during the past 20 to 100 years. Among 42 sites for which recent trend data are available, 10 (24%) are increasing, three (7%) are stable and 29 (69%) are decreasing. Encouragingly, nesting range along Mexico and Central America appears not to have contracted and estimates continue to increase as additional dedicated study is conducted in the eastern Pacific (Gaos *et al.* 2010a).

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

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

Pacific Ocean. American Samoa and Western Samoa host fewer than 30 females annually (Tuato'o-Bartley *et al.* 1993, Grant *et al.* 1997). In Guam, only 5-10 females are estimated to nest annually (G. Balazs, NMFS, in litt. to J. Mortimer 2007; G. Davis, NMFS, in litt. to J. Mortimer 2007) and the same is true for Hawaii, but there are indications that this population is increasing (G. Balazs, pers. comm. in NMFS and USFWS 2007e). Additional populations are known from the eastern Pacific (potentially extending from Mexico through Panama), northeastern Australia, and Malaysia (Hutchinson and Dutton 2007). El Salvador is now known to host the majority of hawksbill turtle nesting activity in the eastern Pacific, with 79.6% (n= 5430) of all nesting observation records, and Mexico hosting the majority of records of hawksbill turtles at sea, with 60.3% (n= 544) of all in-water observation records (Gaos *et al.* 2010b). Total number of nesting females for the Central Pacific hawksbill population was estimated at 940 – 1,200 females annually for the last few years, with an overall downward trend (NMFS and USFWS 2007b).

Indian Ocean. The Indian Ocean hosts several populations of hawksbill sea turtles (Spotila 2004a, Hutchinson and Dutton 2007). These include western Australian, Andaman and Nicobar islands, Maldives, Seychelles, Burma, East Africa, Egypt, Oman, Saudi Arabia, Sudan, and Yemen.

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

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

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

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

Critical habitat. On September 2, 1998, the NMFS established critical habitat for hawksbill sea turtles around Mona and Monito Islands, Puerto Rico (63 FR 46693). Aspects of these areas that are important for hawksbill sea turtle survival and recovery include important natal development habitat, refuge from predation, shelter between foraging periods, and food for hawksbill sea turtle prey.

Kemp's ridley sea turtle

Distribution. The Kemp's ridley was formerly known only from the Gulf of Mexico and along the Atlantic coast of the US (TEWG 2000a). However, recent records support Kemp's ridley sea turtles distribution extending into the Mediterranean Sea on occasion (Tomas and Raga 2008). The vast majority of individuals stem from breeding beaches at Rancho Nuevo on the Gulf of Mexico coast of Mexico.

Movement and migration. Tracking of post-nesting females from Rancho Nuevo and Texas beaches indicates that turtles move along coastal migratory corridors either to the north or south from the nesting beach (Byles 1989b, Byles and Plotkin 1994, Renaud 1995b, Renaud *et al.* 1996, Shaver 1999, 2002). These migratory corridors appear to extend throughout the coastal areas of the Gulf of Mexico and most turtles appear to travel in waters less than roughly 164 feet in depth. Turtles that headed north and east traveled as far as southwest Florida, whereas those that headed south and east traveled as far as the Yucatan Peninsula, Mexico (Morreale *et al.* 2007).

Following migration, Kemp’s ridley sea turtles settle into resident feeding areas for several months (Byles and Plotkin 1994, Morreale *et al.* 2007). Females may begin returning along relatively shallow migratory corridors toward the nesting beach in the winter in order to arrive at the nesting beach by early spring.

Reproduction. Mating is believed to occur about three to four weeks prior to the first nesting (Rostal 2007), or late March through early to mid April. It is presumed that most mating takes place near the nesting beach (Morreale *et al.* 2007, Rostal 2007). Females initially ovulate within a few days after successful mating and lay the first clutch approximately two to four weeks later; if a turtle nests more than once per season, subsequent ovulations occur within approximately 48 hours after each nesting (Rostal 2007).

Approximately 60% of Kemp's ridley nesting occurs along an approximate 25-mile stretch of beach near Rancho Nuevo, Tamaulipas, Mexico from April to July, with limited nesting to the north (100 nests along Texas in 2006) and south (several hundred nests near Tampico, Mexico in 2006 USFWS 2006). Nesting at this location may be particularly important because hatchlings can more easily migrate to foraging grounds (Putman *et al.* 2010). The Kemp's ridley sea turtle tends to nest in large aggregations or arribadas (Bernardo and Plotkin 2007). The period between Kemp's ridley arribadas averages approximately 25 days, but the precise timing of the arribadas is unpredictable (Rostal *et al.* 1997, Bernardo and Plotkin 2007). Like all sea turtles, Kemp's ridley sea turtles nest multiple times in a single nesting season. The most recent analysis suggests approximately 3.075 nests per nesting season per female (Rostal 2007). The annual average number of eggs per nest (clutch size) is 94 to 100 and eggs typically take 45 to 58 days to hatch, depending on temperatures (Marquez-M. 1994, USFWS 2000, 2001, 2002, 2003, 2004, 2005a, 2006, Rostal 2007). The period between nesting seasons for each female is approximately 1.8 to 2.0 years. The nesting beach at Rancho Nuevo may produce a "natural" hatchling sex ratio that is female-biased, which can potentially increase egg production as those turtles reach sexual maturity (Coyne and Landry Jr. 2007, Wibbels 2007).

Growth. Kemp's ridleys require approximately 1.5 to two years to grow from a hatchling to a size of approximately 7.9 inches long, at which size they are capable of making a transition to a benthic coastal immature stage, but can range from one to four years or more. Based on the size of nesting females, it is assumed that turtles must attain a size of approximately 23.6 inches long prior to maturing (Marquez-M. 1994). Growth models based on mark-recapture data suggest that a time period of seven to nine years would be required for this growth from benthic immature to mature size (Schmid and Witzell 1997b, Snover *et al.* 2007b). Currently, age to sexual maturity is believed to range from approximately 10 to 17 years for Kemp's ridleys (Caillouet Jr. *et al.* 1995, Schmid and Witzell 1997a, Snover *et al.* 2007a, Snover *et al.* 2007b). However, estimates of 10 to 13 years predominate in previous studies (Caillouet *et al.* 1995, Schmid and Witzell 1997b, TEWG 2000a).

Habitat. Stranding data indicate that immature turtles in this benthic stage are found in coastal habitats of the entire Gulf of Mexico and US Atlantic coast (TEWG 2000a, Morreale *et al.* 2007). Developmental habitats for juveniles occur throughout the entire coastal Gulf of Mexico and US Atlantic coast northward to New England (Schmid 1998, Wibbels *et al.* 2005, Morreale *et al.* 2007). Key foraging areas in the Gulf of Mexico include Sabine Pass, Texas; Caillou Bay and Calcasieu Pass, Louisiana; Big Gulley, Alabama; Cedar Keys, Florida; and Ten Thousand Islands, Florida. Foraging areas studied along the Atlantic coast include Pamlico Sound, Chesapeake Bay, Long Island Sound, Charleston Harbor, and Delaware Bay. Near-shore waters of 120 feet or less provide the primary marine habitat for adults, although it is not uncommon for adults to venture into deeper waters.

Benthic coastal waters of Louisiana and Texas seem to be preferred foraging areas for Kemp's ridley sea turtles (particularly passes and beachfronts), although individuals may travel along the entire coastal margin of the Gulf of Mexico. Sightings are less frequent during winter and spring, but this is likely due to lesser sighting effort during these times (Shoop and Kenney 1992b, Keinath *et al.* 1996).

Feeding. Kemp's ridley diet consists mainly of swimming crabs, but may also include fish, jellyfish, and an array of mollusks. Immature Kemp's ridleys off southwest Florida documented predation on benthic tunicates, a previously undocumented food source for this species (Witzell and Schmid 2005).

Status and trends. The Kemp's ridley sea turtle was listed as endangered on December 2, 1970 (35 FR 18319). Internationally, the Kemp's ridley is considered the most endangered sea turtle (NRC 1990c, USFWS 1999).

During the mid 20th century, the Kemp's ridley was abundant in the Gulf of Mexico. Historic information indicates that tens of thousands of Kemp's ridleys nested near Rancho Nuevo, Mexico, during the late 1940s (Hildebrand 1963). From 1978 through the 1980s, arribadas were 200 turtles or less, and by 1985, the total number of nests at Rancho Nuevo had dropped to approximately 740 for the entire nesting season, or a projection of roughly 234 turtles (USFWS and NMFS 1992, TEWG 2000a). Beginning in the 1990s, an increasing number of beaches in Mexico were being monitored for nesting, and the total number of nests on all beaches in Tamaulipas and Veracruz in 2002 was over 6,000; the rate of increase from 1985 ranged from 14-16% (TEWG 2000a, USFWS 2002, Heppell *et al.* 2005). In 2006, approximately 7,866 nests were laid at Rancho Nuevo with the total number of nests for all the beaches in Mexico estimated at about 12,000 nests, which amounted to about 4,000 nesting females based upon three nests per female per season (Rostal *et al.* 1997, USFWS 2006, Rostal 2007). Considering remigration rates, the population included approximately 7,000 to 8,000 adult female turtles at that time (Marquez *et al.* 1989, TEWG 2000a, Rostal 2007). Most recently, the 2007 nesting season included an arribada of over 4,000 turtles over a three-day period at Rancho Nuevo (P. Burchfield, pers. comm. in NMFS and USFWS 2007d). The increased recruitment of new adults is illustrated in the proportion of first time nesters, which has increased from 6% in 1981 to 41% in 1994. Average population growth was estimated at 13% per year between 1991 and 1995 (TEWG 1998a). In 2008, there were 17,882 nests in Mexico (Gladys Porter Zoo 2008), and nesting in 2009 reached 21,144 (Gladys Porter Zoo 2010). In 2010, nesting declined significantly, to 13,302 (Gladys Porter Zoo 2010) but it is too early to determine if this is a one-time decline or if is indicative of a change in the trend. Population modelling used by the TEWG (2000b) projected that Kemp's ridleys could reach the recovery plan's intermediate recovery goal of 10,000 nesters by the year 2015. Recent calculations of nesting females determined from nest counts show that the population trend is increasing towards that recovery goal, with an estimate of 4,047 nesters in 2006 and 5,500 in 2007 (NMFS and USFWS 2007c).

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

Natural threats. Sea turtles face predation primarily by sharks and to a lesser extent by killer whales. All sea turtles except leatherbacks can undergo "cold stunning" if water temperatures drop below a threshold level, which can pose lethal effects. Kemp's ridley sea turtles are particularly prone to this phenomenon along Cape Cod (Innis *et al.* 2009). In the last five years (2006-2010), the number of cold-stunned turtles on Cape Cod beaches averaged 115 Kemp's ridleys.

Anthropogenic threats. Population decline has been curtailed due to the virtual elimination of sea turtle and egg harvesting, as well as assistance in hatching and raising hatchlings (head-start). However, habitat destruction remains a concern in the form of bottom trawling and shoreline development. Trawling destroys habitat utilized by Kemp's ridley sea turtles for feeding and construction activities can produce hazardous runoff. Bycatch is also a source of mortality for Kemp's ridley sea turtles (McClellan *et al.* 2009). Finkbeiner *et al.* (2011) estimated that annual bycatch interactions total at least 98,300 individuals annually for US Atlantic fisheries (resulting in 2,700 mortalities or more). The vast majority of fisheries interactions with sea turtles in the US are either Kemp's ridley's or loggerhead sea turtles (Finkbeiner *et al.* 2011).

Toxin burdens in Kemp's ridley sea turtles include DDT, DDE, PCBs, PFOA, PFOS, chlordane, and other organochlorines (Lake *et al.* 1994, Rybitski *et al.* 1995, Keller *et al.* 2004a, Keller *et al.* 2005). These contaminants have the potential to cause deficiencies in endocrine, developmental and reproductive health, and are known to depress immune function in loggerhead sea turtles (Keller *et al.* 2006, Storelli *et al.* 2007b). Along with loggerheads, Kemp's ridley sea turtles have higher levels of PCB and DDT than leatherback and green sea turtles (Pugh and Becker 2001a). Organochlorines, including DDT, DDE, DDD, and PCBs have been identified as bioaccumulative agents and in greatest concentration in subcutaneous lipid tissue (Rybitski *et al.* 1995). Concentrations ranged from 7.46 mu g/kg to 607 mu g/kg, with a mean of 252 mu g/kg in lipid tissue. Five PCB congeners composed most of the contaminants: 153/132, 138/158, 180, 118, and 187 in order of concentration. PCBs have also been identified in the liver, ranging in concentration from 272 ng/g to 655 ng/g of wet weight, values that are several fold higher than in other sea turtle species (Lake *et al.* 1994). However, concentrations are reportedly 5% of that which causes reproductive failure in snapping turtles. DDE was identified to range from 137 ng/g to 386 ng/g wet weight. Trans-nonachlor was found at levels between 129 ng/g and 275 ng/g wet weight. Blood samples may be appropriate proxies for organochlorines in other body tissues (Keller *et al.* 2004a).

Perfluorinated compounds in the forms of PFOA and PFOS have been identified in the blood of Kemp's ridley turtles at concentrations of 39.4 ng/mL and 3.57 ng/mL, respectively (Keller *et al.* 2005). PFCAs have also been detected. It is likely that age and habitat are linked to PFC bioaccumulation.

Oil can also be hazardous to Kemp's ridley turtles, with fresh oil causing significant mortality and morphological changes in hatchlings, but aged oil having no detectable effects (Fritts and McGehee 1981). Blood levels of metals are lower in Kemp's ridley sea turtles than in other sea turtles species or similar to them, with copper (215 ng/g to 1,300 ng/g), lead (0 to 34.3 ng/g), mercury (0.5 ng/g to 67.3 ng/g), silver (0.042 ng/g to 2.74 ng/g), and zinc (3,280 ng/g to 18,900 ng/g) having been identified (Orvik 1997, Innis *et al.* 2008). It is likely that blood samples can be used as an indicator of metal concentration. Mercury has been identified in all turtle species studied, but are generally an order of magnitude lower than toothed whales. The higher level of contaminants found in Kemp's ridley sea turtles are likely due to this species tendency to feed higher on the food chain than other sea turtles. Females from sexual maturity through reproductive life should have lower levels of contaminants than males because contaminants are shared with progeny through egg formation.

Critical habitat. NMFS has not designated critical habitat for Kemp's ridley sea turtle.

Green sea turtle

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

Population designation. Populations are distinguished generally by ocean basin and more specifically by nesting location (Table 6).
Based upon genetic differences, two or three distinct regional clades may exist in the Pacific: western Pacific and South Pacific islands, eastern Pacific, and central Pacific, including the rookery at French Frigate Shoals, Hawaii (Dutton *et al.* 1996, Dutton and Balazs In review). In the eastern Pacific, green sea turtles forage from San Diego Bay, California to Mejillones, Chile. Individuals along the southern foraging area originate from Galapagos Islands nesting beaches, while those in the Gulf of California originate primarily from Michoacán. Green turtles foraging in San Diego Bay and along the Pacific coast of Baja California originate primarily from rookeries of the Islas Revillagigedos (Dutton 2003a).

Table 6. Locations and most recent abundance estimates of threatened green sea turtles as annual nesting females (AF), annual nests (AN), annual egg production (EP), and annual egg harvest (EH).

Location	Most recent abundance	Reference
Western Atlantic Ocean		
Tortuguero, Costa Rica	17,402-37,290 AF	(Troëng and Rankin 2005)
Aves Island, Venezuela	335-443 AF	(Vera 2007)
Galibi Reserve, Suriname	1,803 AF	(Weijerman *et al.* 1998)
Isla Trindade, Brazil	1,500-2,000 AF	(Moreira and Bjorndal 2006)
Central Atlantic Ocean		
Ascension Island, UK	3,500 AF	(Broderick *et al.* 2006)
Eastern Atlantic Ocean		
Poilao Island, Guinea-Bissau	7,000-29,000 AN	(Catry *et al.* 2009)
Bioko Island, Equatorial Guinea	1,255-1,681 AN	(Tomas *et al.* 1999)
Mediterranean Sea		

Location	Most recent abundance	Reference
Turkey	214-231 AF	(Broderick *et al.* 2002)
Cyprus	121-127 AF	(Broderick *et al.* 2002)
Israel / Palestine	1-3 AF	(Kuller 1999)
Syria	100 AN	(Rees *et al.* 2005)
Western Indian Ocean		
Eparces Islands	2,000-11,000 AF	(Le Gall *et al.* 1986)
Comoros Islands	5,000 AF	S. Ahamada, pers. comm. 2001
Seychelles Islands	3,535-4,755 AF	J. Mortimer, pers. comm. 2002
Kenya	200-300 AF	(Okemwa and Wamukota 2006)
Northern Indian Ocean		
Ras al Hadd, Oman	44,000 AN	S. Al-Saady, pers. comm. 2007
Sharma, Yemen	15 AF	(Saad 1999)
Karan Island, Saudi Arabia	408-559 AF	(Pilcher 2000)
Jana and Juraid Islands, Saudi Arabia	643 AN	(Pilcher 2000)
Hawkes Bay and Sandspit, Pakistan	600 AN	(Asrar 1999)
Gujarat, India	461 AN	(Sunderraj *et al.* 2006)
Sri Lanka	184 AF	(Kapurisinghe 2006)
Eastern Indian Ocean		
Thamihla Kyun, Myanmar	<250,000 EH	(Thorbjarnarson *et al.* 2000)
Pangumbahan, Indonesia	400,000 EH	(Schulz 1987)
Suka Made, Indonesia	395 AN	C. Limpus, pers. comm. 2002
Western Australia	3,000-30,000 AN	R. Prince, pers. comm. 2001
Southeast Asia		
Gulf of Thailand	250 AN	Charuchinda pers. comm. 2001
Vietnam	239 AF	(Hamann *et al.* 2006b)
Berau Islands, Indonesia	4,000-5,000 AF	(Schulz 1984)
Turtle Islands, Philippines	1.4 million EP	(Cruz 2002)
Sabah Turtle Islands, Malaysia	8,000 AN	(Chan 2006)
Sipadan, Malaysia	800 AN	(Chan 2006)
Sarawak, Malaysia	2,000 AN	(Liew 2002)
Enu Island (Aru Islands)	540 AF	Dethmers, in preparation
Terengganu, Malaysia	2,200 AN	(Chan 2006)
Western Pacific Ocean		
Heron Island and southern Great Barrier Reef areas, Australia	5,000-10,000 AF	(Maison *et al.* 2010)
Raine Island and northern Great Barrier Reef areas, Australia	10,000-25,000 AF	(Limpus *et al.* 2003) (Maison *et al.* 2010)
Coringa-Herald National Nature Reserve, Australia	1,445 AF	(Maison *et al.* 2010)
Guam	45 AF	(Cummings 2002)
Phoenix Islands, Kiribati	100-300 AF	(Maison *et al.* 2010)
Ogasawara Islands, Japan	500 AF	(Chaloupka *et al.* 2007)
Micronesia	500-1,000 AF	(Maison *et al.* 2010)
Marshall Islands	100-500 AF	(Maison *et al.* 2010)
New Caledonia	1,000-2,000 AF	(Maison *et al.* 2010)
Central and Eastern Pacific Ocean		
French Frigate Shoals, Hawaii	400 AF	(Balazs and Chaloupka 2006)
Michoacán, Mexico	1,395 AF	C. Delgado, pers. comm. 2006
Central American Coast	184-344 AN	(López and Arauz 2003)
Galapagos Islands, Ecuador	1,650 AF	(Zárate *et al.* 2006)

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

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

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

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

Migration and movement. Green sea turtles are highly mobile and undertake complex movements through geographically disparate habitats during their lifetimes (Musick and Limpus 1997, Plotkin 2003). The periodic migration between nesting sites and foraging areas by adults is a prominent feature of their life history. After departing as hatchlings and residing in a variety of marine habitats for 40 or more years (Limpus and Chaloupka 1997), green sea turtles make their way back to the same beach from which they hatched (Carr *et al.* 1978, Meylan *et al.* 1990). At approximately 20-25 cm carapace length, juveniles leave pelagic habitats and enter benthic foraging areas (Bjorndal 1997a). Green sea turtles spend the majority of their lives in coastal foraging grounds. These areas include both open coastline and protected bays and lagoons. While in these areas, green sea turtles rely on marine algae and seagrass as their primary dietary constituents, although some populations also forage heavily on invertebrates. There is some evidence that individuals move from shallow seagrass beds during the day to deeper areas at night (Hazel 2009). However, avoidance of areas of greater than 10 m when moderate depths of 5-10 m with sea grass beds has been found, with speed and displacement from capture locations being similar at night as during the daytime (Senko *et al.* 2010a).

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

Green sea turtles in the Gulf of Mexico tend to remain along the coast (lagoons, channels, inlets, and bays), with nesting primarily occurring in Florida and Mexico and infrequent nesting in all other areas (NMFS and USFWS 1991, Meylan *et al.* 1995a, USAF 1996, Landry and Costa 1999). Foraging areas seem to be based upon seagrass and macroalgae abundance, such as in the Laguna Madre of Texas. However, green sea turtles may also occur in offshore regions, particularly during migration and development.

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

Status and trends. Federal listing of the green sea turtle occurred on July 28, 1978, with all populations listed as threatened except for the Florida and Pacific coast of Mexico breeding populations, which are endangered (43 FR 32800). The International Union for Conservation of Nature (IUCN) has classified the green turtle as "endangered."

No trend data are available for almost half of the important nesting sites, where numbers are based on recent trends and do not span a full green sea turtle generation, and impacts occurring over four decades ago that caused a change in juvenile recruitment rates may have yet to be manifested as a change in nesting abundance. The numbers also only reflect one segment of the population (nesting females), who are the only segment of the population for which reasonably good data are available and are cautiously used as one measure of the possible trend of populations.

Table 12 summarizes nesting abundance for 46 nesting sites worldwide. These include both large and small rookeries believed to be representative of the overall trends for their respective regions. Based on the mean annual reproductive effort, 108,761-150,521 females nest each year among the 46 sites. Overall, of the 26 sites for which data enable an assessment of current trends, 12 nesting populations are increasing, 10 are stable, and four are decreasing. Long-term continuous datasets of 20 years are available for 11 sites, all of which are either increasing or stable. Despite the apparent global increase in numbers, the positive overall trend should be viewed cautiously because trend data are available for just over half of all sites examined and very few data sets span a full green sea turtle generation (Seminoff 2004b).

Atlantic Ocean. Primary sites for green sea turtle nesting in the Atlantic/Caribbean include: (1) Yucatán Peninsula, Mexico; (2) Tortuguero, Costa Rica; (3) Aves Island, Venezuela; (4) Galibi Reserve, Suriname; (5) Isla Trindade, Brazil; (6) Ascension Island, United Kingdom; (7) Bioko Island, Equatorial Guinea; and (8) Bijagos Achipelago, Guinea-Bissau (NMFS and USFWS 2007a). Nesting at all of these sites was considered to be stable or increasing with the exception of Bioko Island and the Bijagos Archipelago where the lack of sufficient data precludes a meaningful trend assessment for either site (NMFS and USFWS 2007a). Seminoff (2004a) reviewed green sea turtle nesting data for eight sites in the western, eastern, and central Atlantic. Seminoff (2004a) concluded that all sites in the central and western Atlantic showed increased nesting, with the exception of nesting at Aves Island, Venezuela, while both sites in the eastern Atlantic demonstrated decreased nesting. These sites are not inclusive of all green sea turtle nesting in the Atlantic. However, other sites are not believed to support nesting levels high enough that would change the overall status of the species in the Atlantic (NMFS and USFWS 2007a).

By far, the most important nesting concentration for green sea turtles in the western Atlantic is in Tortuguero, Costa Rica (NMFS and USFWS 2007a). Nesting in the area has increased considerably since the 1970s and nest count data from 1999-2003 suggest nesting by 17,402-37,290 females per year (NMFS and USFWS 2007a). The number of females nesting per year on beaches in the Yucatán, at Aves Island, Galibi Reserve, and Isla Trindade number in the hundreds to low thousands, depending on the site (NMFS and USFWS 2007a).

The vast majority of green sea turtle nesting within the southeastern US occurs in Florida (Johnson and Ehrhart 1994, Meylan *et al.* 1995b). Green sea turtle nesting in Florida has been increasing since 1989 (Florida Fish and Wildlife Conservation Commission, Florida Marine Research Institute Index Nesting Beach Survey Database). Since establishment of index beaches in 1989, the pattern of green turtle nesting shows biennial peaks in abundance with a generally positive trend during the ten years of regular monitoring. This is perhaps due to increased protective legislation throughout the Caribbean (Meylan *et al.* 1995b). A total statewide average (all beaches, including index beaches) of 5,039 green turtle nests were laid annually in Florida between 2001 and 2006, with a low of 581 in 2001 and a high of 9,644 in 2005 (NMFS and USFWS 2007a). Data from index nesting beaches substantiate the dramatic increase in nesting. In 2007, there were 9,455 green turtle nests found just on index nesting beaches, the highest since index beach monitoring began in 1989. The number fell back to 6,385 in 2008, further dropping under 3,000 in 2009, but that consecutive drop was a temporary deviation from the normal biennial nesting cycle for green turtles, as 2010 saw an increase back to 8,426 nests on the index nesting beaches (FWC Index Nesting Beach Survey Database). Occasional nesting has been documented along the Gulf coast of Florida (Meylan *et al.* 1995b). More recently, green turtle nesting occurred on Bald Head Island, North Carolina; just east of the mouth of the Cape Fear River; on Onslow Island; and on Cape Hatteras National Seashore. In 2010, a total of 18 nests were found in North Carolina, 6 nests in South Carolina, and 6 nests in Georgia (nesting databases maintained on www.seaturtle.org). Increased nesting has also been observed along the Atlantic coast of Florida, on beaches where only loggerhead nesting was observed in the past (Pritchard 1997). Recent modeling by Chaloupka *et al.* (2008a)using data sets of 25 years or more has resulted in an estimate of the Florida nesting stock at the Archie Carr National Wildlife Refuge growing at an annual rate of 13.9%, and the Tortuguero, Costa Rica, population growing at 4.9%.

There are no reliable estimates of the number of immature green sea turtles that inhabit coastal areas of the southeastern US. However, information on incidental captures of immature green sea turtles at the St. Lucie Power Plant in St. Lucie County, Florida, shows that the annual number of immature green sea turtles captured by their offshore cooling water intake structures has increased significantly. Green sea turtle annual captures averaged 19 for 1977-1986, 178 for 1987-1996, and 262 for 1997-2001 (Florida Power and Light Company St. Lucie Plant 2002). More recent unpublished data shows 101 captures in 2007, 299 in 2008, 38 in 2009 (power output was cut—and cooling water intake concomitantly reduced—for part of that year) and 413 in 2010. Ehrhart *et al.* (2007) documented a significant increase in in-water abundance of green turtles in the Indian River Lagoon area.

Pacific Ocean. Green turtles are thought to be declining throughout the Pacific Ocean, with the exception of Hawaii, from a combination of overexploitation and habitat loss (Eckert 1993b, Seminoff *et al.* 2002a). In the western Pacific, the only major (>2,000 nesting females) populations of green turtles occur in Australia and Malaysia, with smaller colonies throughout the area. Indonesian nesting is widely distributed, but has experienced large declines over the past 50 years. Hawaii green turtles are genetically distinct and geographically isolated, and the population appears to be increasing in size despite the prevalence of fibropapillomatosis and spirochidiasis (Aguirre *et al.* 1998).

The East Island nesting beach in Hawaii is showing a 5.7% annual growth rate over >25 years (Chaloupka *et al.* 2008a). In the Eastern Pacific, mitochondrial DNA analysis has indicated three key nesting populations: Michoacán, Mexico; Galapagos Islands, Ecuador; and Islas Revillagigedos, Mexico (Dutton 2003b). The number of nesting females per year exceeds 1,000 females at each site (NMFS and USFWS 2007a). However, historically, >20,000 females per year are believed to have nested in Michoacán alone (Clifton *et al.* 1982, NMFS and USFWS 2007a). Thus, the current number of nesting females is still far below historical levels. Datasets over 25 years in Chichi-jima, Japan; Heron Island, Australia; and Raine Island, Australia, show increases in abundance (Chaloupka *et al.* 2008a).

Indian Ocean. One of the largest nesting sites for green sea turtles worldwide occurs on the beaches of Oman where an estimated 20,000 green sea turtles nest annually (Hirth 1997a). Only the Comoros Island index site in the western Indian Ocean showed evidence of increased nesting (Seminoff 2004a).

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

Anthropogenic threats. Major anthropogenic impacts to the nesting and marine environment affect green sea turtle survival and recovery. At nesting beaches, green sea turtles rely on intact dune structures, native vegetation, and normal beach temperatures for nesting (Ackerman 1997). Structural impacts to nesting habitat include the construction of buildings and pilings, beach armoring and renourishment, and sand extraction (Lutcavage *et al.* 1997b, Bouchard *et al.* 1998). These factors may directly, through loss of beach habitat, or indirectly, through changing thermal profiles and increasing erosion, serve to decrease the amount of nesting area available to nesting females, and may evoke a change in the natural behaviors of adults and hatchlings (Ackerman 1997, Witherington *et al.* 2003, 2007). On the Pacific coast of Mexico in the mid-1970s, >70,000 green turtle eggs were harvested every night. The presence of lights on or adjacent to nesting beaches alters the behavior of nesting adults (Witherington 1992) and is often fatal to emerging hatchlings as they are attracted to light sources and drawn away from the water

(Witherington and Bjorndal 1991). In addition to impacting the terrestrial zone, anthropogenic disturbances also threaten coastal marine habitats, particularly areas rich in seagrass and marine algae. These impacts include contamination from herbicides, pesticides, oil spills, and other chemicals, as well as structural degradation from excessive boat anchoring and dredging (Francour *et al.* 1999, Lee Long *et al.* 2000, Waycott *et al.* 2005). Ingestion of plastic and other marine debris is another source of morbidity and mortality (Stamper *et al.* 2009). Green sea turtles stranded in Brazil were all found to have ingested plastics or fishing debris (n=34), although mortality appears to have results in three cases (Tourinho *et al.* 2009). Low-level bycatch has also been documented in longline fisheries (Petersen *et al.* 2009). Further, the introduction of alien algae species threatens the stability of some coastal ecosystems and may lead to the elimination of preferred dietary species of green sea turtles (De Weede 1996). Very few green sea turtles are bycaught in US fisheries (Finkbeiner *et al.* 2011). However, a legal fishery operates in Madagascar that harvested about 10,000 green turtles annually in the mid-1990s.

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

Green sea turtles have been found to contain the organochlorines chlordane, lindane, endrin, endosulfan, dieldrin, DDT and PCB (Miao *et al.* 2001, Gardner *et al.* 2003). Levels of PCBs found in eggs are considered far higher than what is fit for human consumption (van de Merwe *et al.* 2009). The heavy metals copper, lead, manganese, cadmium, and nickel have also been found in various tissues and life stages (Barbieri 2009). Arsenic also occurs in very high levels in green sea turtle eggs (van de Merwe *et al.* 2009). These contaminants have the potential to cause deficiencies in endocrine, developmental, and reproductive health, and depress immune function in loggerhead sea turtles (Keller *et al.* 2006; Storelli *et al.* 2007). Exposure to sewage effluent may also result in green sea turtle eggs harboring antibiotic-resistant strains of bacteria (Al-Bahry *et al.* 2009). DDE has not been found to influence sex determination at levels below cytotoxicity (Podreka *et al.* 1998, Keller and McClellan-Green 2004). To date, no tie has been found between pesticide concentration and susceptibility to fibropapillomatosis, although degraded habitat and pollution have been tied to the incidence of the disease (Aguirre *et al.* 1994, Foley *et al.* 2005). Flame retardants have been measured from healthy individuals (Hermanussen *et al.* 2008). It has been theorized that exposure to tumor-promoting compounds produced by the cyanobacteria *Lyngbya majuscule* could promote the development of fibropapillomatosis (Arthur *et al.* 2008). It has also been theorized that dinoflagellates of the genus *Prorocentrum* that produce the tumorogenic compound okadoic acid may influence the development of fibropapillomatosis (Landsberg *et al.* 1999).

Critical habitat. On September 2, 1998, critical habitat for green sea turtles was designated in coastal waters surrounding Culebra Island, Puerto Rico (63 FR 46693). Aspects of these areas that are important for green sea turtle survival and recovery include important natal development habitat, refuge from predation, shelter between foraging periods, and food for green sea turtle prey.

Loggerhead sea turtle

Distribution. Loggerheads are circumglobal occurring throughout the temperate and tropical regions of the Atlantic, Pacific, and Indian oceans. Loggerheads are the most abundant species of sea turtle found in US coastal waters.

Population designations. Five groupings represent loggerhead sea turtles by major sea or ocean basin: Atlantic, Pacific, and Indian oceans, as well as Caribbean and Mediterranean seas. As with other sea turtles, populations are frequently divided by nesting aggregation (Hutchinson and Dutton 2007). On March 16, 2010, the NMFS proposed to designate nine distinct population segments (DPSs) of loggerhead sea turtles: South Atlantic Ocean and southwest Indian Ocean as threatened as well as Mediterranean Sea, North Indian Ocean, North Pacific Ocean, northeast Atlantic Ocean, northwest Atlantic Ocean, South Pacific Ocean, and southeast Indo-Pacific Ocean as endangered (75 FR 12598).

Atlantic Ocean. Western Atlantic nesting locations include The Bahamas, Brazil, and numerous locations from the Yucatán Peninsula to North Carolina (Addison and Morford 1996, Addison 1997, Marcovaldi and Chaloupka 2007). This group comprises five nesting subpopulations: Northern, Southern, Dry Tortugas, Florida Panhandle, and Yucatán. Additional nesting occurs on Cay Sal Bank (Bahamas), Cuba, the Bahamian Archipelago, Quintana Roo (Yucatan Peninsula), Colombia, Brazil, Caribbean Central America, Venezuela, and the eastern Caribbean Islands. Genetic studies indicate that, although females routinely return to natal beaches, males may breed with females from multiple populations and facilitate gene flow Bowen *et al.* (2005). In the eastern Atlantic, we know of five rookeries from Cape Verde, Greece, Libya, Turkey, and the western Africa coast.

Indian Ocean. Loggerhead sea turtles nest along the Indian Ocean in Oman, Yemen, Sri Lanka, Madagascar, South Africa, and possibly Mozambique.

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

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

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

Loggerhead mating likely occurs along migration routes to nesting beaches, as well as in offshore from nesting beaches several weeks prior to the onset of nesting (Dodd 1988a, NMFS and USFWS 1998d). Females usually breed every 2-3 years, but can vary from 1-7 years (Richardson *et al.* 1978, Dodd 1988a). Females lay an average of 4.1 nests per season (Murphy and Hopkins 1984) , although recent satellite telemetry from nesting females along southwest Florida support 5.4 nests per female per season, with increasing numbers of eggs per nest during the course of the season (Tucker 2009). The authors suggest that this finding warrants revision of the number of females nesting in the region. The western Atlantic breeding season is March-August. Nesting sites appear to be related to beaches with relatively high exposure to wind or wind-generated waves (Santana Garcon *et al.* 2010).

The Japanese rookeries are the most significant nesting sites for loggerheads in the North Pacific, with nesting occurring on the Japanese mainland, except for Hokkaido, as well as the Ryukyu Islands to the south (Kamezaki 1989, Uchida and Nishiwaki 1995, Kamezaki *et al.* 2003, Sea Turtle Association of Japan 2010). Nesting generally occurs through summer and fall (April-August, peaking in July), with females returning every two to three years (Iwamoto *et al.* 1985). Nesting females lay at least three nests of 60-115 eggs per nest each season, with roughly two weeks between nests (Iwamoto *et al.* 1985, Eckert 1993b, Nishimura 1994). Between nests, females appear to swim offshore into the Kuroshio Current, possibly to speed egg development (NMFS and USFWS 1998b, Sato *et al.* 1998).

Nesting in the Gulf of Mexico does occur, although primarily in Florida, with rare nests do occur along North and South Padre Island in Texas (Hildebrand 1983, Dodd 1988b).

Migration and movement. Loggerhead hatchlings migrate offshore and become associated with *Sargassum* spp. habitats, driftlines, and other convergence zones (Carr 1986). After 14-32 years of age, they shift to a benthic habitat, where immature individuals forage in the open ocean and coastal areas along continental shelves, bays, lagoons, and estuaries (NMFS 2001b, Bowen *et al.* 2004). Adult loggerheads make lengthy migrations from nesting beaches to foraging grounds (TEWG 1998b). In the Gulf of Mexico, larger females tend to disperse more broadly after nesting than smaller individuals, which tend to stay closer the nesting location (Girard *et al.* 2009). In the North Atlantic, loggerheads travel north during spring and summer as water temperatures warm and return south in fall and winter, but occur offshore year-round assuming adequate temperature. For immature individuals, this movement occurs in two patterns: a north-south movement over the continental shelf with migration south of Cape Hatteras in winter and movement north along Virginia for summer foraging, and a not-so-seasonal oceanic dispersal into the Gulf Stream as far north as the 10-15˚ C isotherm (Mansfield *et al.* 2009). Wallace *et al.* (2009) suggested differences in growth rate based upon these foraging strategies. There is conflicting evidence that immature loggerheads roam the oceans in currents and eddies and mix from different natal origins or distribute on a latitudinal basis that corresponds with their natal beaches (Monzon-Arguello *et al.* 2009, Wallace *et al.* 2009). McCarthy *et al.* (2010) found that movement patterns of loggerhead sea turtles were more convoluted when sea surface temperatures were higher, ocean depths shallower, ocean currents stronger, and chlorophyll a levels lower.

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

Sighting and stranding records support loggerhead sea turtles to be common, year-round residents of the Gulf of Mexico, although their abundance is much greater in the northeastern region versus the northwestern (Fritts *et al.* 1983, Landry and Costa 1999, Davis *et al.* 2000). Loggerheads may occur in both offshore habitats (particularly around oil platforms and reefs, where prey and shelter are available; (Fritts *et al.* 1983, Rosman *et al.* 1987, Lohoefener *et al.* 1990, Gitschlag and Herczeg 1994, Davis *et al.* 2000), as well as shallow bays and sounds (which may be important developmental habitat for late juveniles in the eastern Gulf of Mexico;

(Lohoefener *et al.* 1990, USAF 1996, Davis *et al.* 2000). Offshore abundance in continental slope waters increases during the winter in the eastern Gulf of Mexico, as cooler inshore waters force individuals into warmer offshore areas (Davis *et al.* 2000).

Gender, age, and survivorship. Although information on males is limited, several studies identified a female bias, although a single study has found a strong male bias to be possible (Dodd 1988a, NMFS 2001b, Rees and Margaritoulis 2004).

Additionally, little is known about longevity, although Dodd (1988a) estimated the maximum female life span at 47-62 years. Heppell *et al.* (2003a) estimated annual survivorship to be 0.81 (southeast US adult females), 0.78-0.91 (Australia adult females), 0.68-0.89 (southeast US benthic juveniles, and 0.92 (Australia benthic juveniles). Survival rates for hatchlings during their first year are likely very low (Heppell *et al.* 2003a, Heppell *et al.* 2003b).

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

Status and trends. Loggerhead sea turtles were listed as threatened under the ESA of 1973 on July 28, 1978 (43 FR 32800). The NMFS recently determined that a petition to reclassify loggerhead turtles in the western North Atlantic Ocean as endangered may be warranted due to the substantial scientific and commercial information presented. Consequently, NMFS has initiated a review of the status of the species and is currently soliciting additional information on the species status and ecology, as well as areas that may qualify as critical habitat (73 FR 11849; March 5, 2008).

There is general agreement that the number of nesting females provides a useful index of the species’ population size and stability at this life stage, even though there are doubts about the ability to estimate the overall population size (Bjorndal *et al.* 2005). An important caveat for population trends analysis based on nesting beach data is that this may reflect trends in adult nesting females, but it may not reflect overall population growth rates well. Adult nesting females often account for less than 1% of total population numbers. The global abundance of nesting female loggerhead turtles is estimated at 43,320–44,560 (Spotila 2004b).

Atlantic Ocean. In the eastern Atlantic, the Cape Verde Islands support the only known loggerhead nesting assemblage, which is of at least intermediate size (Fretey 2001a); 1,071 nests were observed in 2009 (Lino *et al.* 2010). In 2000, researchers tagged over 1,000 nesting females (Erhart *et al.* 2003). Annual data from monitoring projects in Cyprus, Greece, Israel, Tunisia, and Turkey reveal total annual nesting in the Mediterranean ranging of 3,375-7,085 nests per season (Margaritoulis *et al.* 2003). Libya and the West African coast host genetically-unique breeding populations of loggerhead sea turtles as well (Hutchinson and Dutton 2007). A recently discovered nesting site along the southern Italian shores of the Ionian Sea found particularly high genetic diversity amongst nesting females (Garofalo *et al.* 2009). Nesting at Dalyan Beach, Turkey does not have an apparent trend, with between 50 and 286 nests laid annually for the past 19 years (Turkozan and Yilmaz 2008).

The greatest concentration of loggerheads occurs in the Atlantic Ocean and the adjacent Caribbean Sea, primarily on the Atlantic coast of Florida, with other major nesting areas located on the Yucatán Peninsula of Mexico, Columbia, Cuba, South Africa (Márquez 1990, EuroTurtle 2006 as cited in LGL Ltd. 2007).

Among the five subpopulations, loggerhead females lay 53,000-92,000 nests per year in the southeastern US and the Gulf of Mexico, and the total number of nesting females is 32,000-56,000. All of these are currently in decline or data are insufficient to access trends (TEWG 1998a, NMFS 2001b). Loggerheads from western North Atlantic nesting aggregations may or may not feed in the same regions from which they hatch. Loggerhead sea turtles from the northern nesting aggregation, which represents about 9% of the loggerhead nests in the western North Atlantic, comprise 25-59% of individuals foraging from Georgia up to the northeast US (Sears 1994, Norrgard 1995, Sears *et al.* 1995, Rankin-Baransky 1997, Bass *et al.* 1998). Loggerheads associated with the South Florida nesting aggregation occur in higher frequencies in the Gulf of Mexico (where they represent ~10% of the loggerhead captures) and the Mediterranean Sea (where they represent ~45% of loggerhead sea turtles captured). About 4,000 nests per year are laid along the Brazilian coast (Ehrhart *et al.* 2003).

The northern recovery unit along Georgia, South Carolina, and North Carolina has a forty-year time-series trend showing an overall decline in nesting, but the shorter comprehensive survey data (20 years) indicate a stable population (GDNR, NCWRC, and SCDNR nesting data located at www.seaturtle.org). NMFS scientists have estimated that the northern subpopulation produces 65% males (NMFS 2001b).

The peninsular Florida recovery unit is the largest loggerhead nesting assemblage in the northwest Atlantic. A near-complete nest census (all beaches including index nesting beaches) undertaken from 1989 to 2007 showed a mean of 64,513 loggerhead nests per year, representing approximately 15,735 nesting females annually (NMFS and USFWS 2008). The statewide estimated total for 2010 was 73,702 (FWRI nesting database). An analysis of index nesting beach data shows a 26% nesting decline between 1989 and 2008, and a mean annual rate of decline of 1.6% despite a large increase in nesting for 2008, to 38,643 nests (FWRI nesting database)(NMFS and USFWS 2008, Witherington *et al.* 2009). In 2009, nesting levels, while still higher than the lows of 2004, 2006, and 2007, dropped below 2008 levels to approximately 32,717 nests, but in 2010 a large increase was seen, with 47,880 nests on the index nesting beaches (FWRI nesting database). The 2010 index nesting number is the largest since 2000. With the addition of data through 2010, the nesting trend for the nowrthwestern Atlantic DPS is slightly negative and not statistically different from zero (no trend)(NMFS and USFWS 2010).

Preliminary, unofficial reports indicate that 2011 nesting may be a high nesting year on par with 2010.

Because of its size, the south Florida subpopulation of loggerheads may be critical to the survival of the species in the Atlantic, and in the past it was considered second in size only to the Oman nesting aggregation (NMFS and USFWS 1991b, NMFS 2006e). The South Florida population increased at ~5.3% per year from 1978-1990, and was initially increasing at 3.9-4.2% after 1990. An analysis of nesting data from 1989-2005, a period of more consistent and accurate surveys than in previous years, showed a detectable trend and, more recently (1998-2005), has shown evidence of a declining trend of approximately 22.3% (FFWCC 2007b, a, Witherington *et al.* 2009). This is likely due to a decline in the number of nesting females within the population (Witherington *et al.* 2009). Nesting data from the Archie Carr Refuge (one of the most important nesting locations in southeast Florida) over the last 6 years shows nests declined from approximately 17,629 in 1998 to 7,599 in 2004, also suggesting a decrease in population size[1]. Loggerhead nesting is thought to consist of just 60 nesting females in the Caribbean and Gulf of Mexico (NMFS 2006f). Based upon the small sizes of almost all nesting aggregations in the Atlantic, the large numbers of individuals killed in fisheries, and the decline of the only large nesting aggregation, we suspect that the extinction probabilities of loggerhead sea turtle populations in the Atlantic are only slightly lower than those of populations in the Pacific.

Zurita *et al.* (2003) found a statistically significant increase in the number of nests on seven of the beaches on Quintana Roo, Mexico, from 1987-2001, where survey effort was consistent during the period. However, nesting has declined since 2001, and the previously reported increasing trend appears to have been temporary (NMFS and USFWS 2008).

Mediterranean Sea. Nesting in the Mediterranean is confined almost exclusively to the eastern basin. The highest level of nesting in the Mediterranean occurs in Greece, with an average of 3,050 nests per year.

Pacific Ocean. Abundance has declined dramatically over the past 10-20 years, although loggerheads range widely from Alaska to Chile (NMFS and USFWS 1998d). Pacific nesting is limited to two major locations, Australia and Japan, although low level nesting may occur outside of Japan in areas surrounding the South China Sea (Kamezaki *et al.* 2003, Chan *et al.* 2007). Eastern Australia supported one of the major global loggerhead nesting assemblages until recently (Limpus 1985). Now, less than 500 females nest annually, an 86% reduction in the size of the annual nesting population in 23 years (Limpus and Limpus 2003). The status of loggerhead nesting colonies in southern Japan and the surrounding region is uncertain, but approximately 1,000 female loggerhead turtles may nest there; a 50-90% decline compared to historical estimates (Dodd Jr. 1988b, Bolten *et al.* 1996, STAJ 2002, Kamezaki *et al.* 2003). Nine major nesting beaches (greater than 100 nests per season) and six “submajor” beaches (10–100 nests per season) exists, including Yakushima island where 40% of nesting occurs (Kamezaki *et al.* 2003). Nesting declined from an initial peak of approximately 6,638 nests in

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[1] While this is a long period of decline relative to the past observed nesting pattern at this location, aberrant ocean surface temperatures complicate the analysis and interpretation of these data. Although caution is warranted in interpreting the decreasing nesting trend given inherent annual fluctuations in nesting and the short time period over which the decline has been noted, the recent nesting decline at this nesting beach is reason for concern.

1990–1991, followed by a steep decline to a low of 2,064 nests in 1997 (Conant *et al.* 2009). During the past decade, nesting increased gradually to 5,167 nests in 2005 (Conant *et al.* 2009), declined and then rose again to a record high of 11,082 nests in 2008, and then 7,495 and 10,121 nests in 2009 and 2010, respectively (STAJ 2008, 2009, 2010).

In addition, loggerheads uncommonly occur in US Pacific waters, and there have been no documented strandings of loggerheads on the Hawaiian Islands in nearly 20 years (1982-1999 stranding data). There are very few records of loggerheads nesting on any of the many islands of the central Pacific, and the species is considered rare or vagrant in this region (USFWS 1998). Overall, Gilman (2009) estimated that the number of loggerheads nesting the Pacific has declined by 80% in the past 20 years.

Indian Ocean. The largest known nesting aggregation occurs on Masirah and Kuria Muria Islands in Oman (Ross and Barwani 1982). Extrapolations resulting from partial surveys and tagging in 1977-1978 provided broad estimates of 19,000-60,000 females nesting annually at Masirah Island, while a more recent partial survey in 1991 provided an estimate of 23,000 nesting females (Ross 1979, Ross and Barwani 1982, Baldwin 1992, Ross 1998). Over 3,000 nests per year have been recorded on the Al-Halaniyat Islands, while along the Oman mainland of the Arabian Sea, about 2,000 nests are deposited per year (Salm 1991, Salm *et al.* 1993). Based upon genetic analyses, additional populations nest in Yemen, Sri Lanka, and Madagascar (Hutchinson and Dutton 2007). In the southwestern Indian Ocean, the highest concentration of nesting occurs on the coast of Tongaland, South Africa (Baldwin *et al.* 2003). The total number of females nesting annually in South Africa is estimated to be between 500-2,000 (Baldwin *et al.* 2003). In the Eastern Indian Ocean, all known nesting sites are found in Western Australia (Dodd Jr. 1988a). An estimated 800-1,500 loggerheads nest annually on Dirk Hartog Island beaches along Western Australia (Baldwin *et al.* 2003).

Natural threats. Sea turtles face predation primarily by sharks and to a lesser extent by killer whales. All sea turtles except leatherbacks can undergo "cold stunning" if water temperatures drop below a threshold level, which can pose lethal effects. In January 2010, an unusually large cold-stunning event occurred throughout the southeast US, with well over 3,000 sea turtles (mostly greens but also hundreds of loggerheads) found cold-stunned. Most survived, but several hundred were found dead or died after being discovered in a cold-stunned state. Eggs are commonly eaten by raccoons and ghost crabs along the eastern US (Barton and Roth 2008). In the water, hatchlings are hunted by herons, gulls, dogfish, and sharks. Heavy loads of barnacles are associated with unhealthy or dead stranded loggerheads (Deem *et al.* 2009).

Anthropogenic threats. Anthropogenic threats impacting loggerhead nesting habitat are numerous: coastal development and construction, placement of erosion control structures, beachfront lighting, vehicular and pedestrian traffic, sand extraction, beach erosion, beach nourishment, beach pollution, removal of native vegetation, and planting of non-native vegetation (Baldwin 1992, USFWS 1998, Margaritoulis *et al.* 2003, Mazaris *et al.* 2009b). Surprisingly, beach nourishment also hampers nesting success, but only in the first year post-nourishment before hatching success increases (Brock *et al.* 2009). Loggerhead sea turtles face numerous threats in the marine environment as well, including oil and gas exploration, marine pollution, trawl, purse seine, hook and line, gill net, pound net, longline, and trap fisheries, underwater explosions, dredging, offshore artificial lighting, power plant entrapment, entanglement in debris, ingestion of marine debris, marina and dock construction and operation, boat collisions, and poaching. At least in the Mediterannean Sea, Anthorpogenic threats appear to disproportionally impact larger (more fecund) loggerheads (Bellido *et al.* 2010).

The major factors inhibiting their recovery include mortalities caused by fishery interactions and degradation of the beaches on which they nest. Shrimp trawl fisheries account for the highest number of captured and killed loggerhead sea turtles. Along the Atlantic coast of the US, the NMFS estimated that shrimp trawls capture almost 163,000 loggerhead sea turtles each year in the Gulf of Mexico, of which 3,948 die. However, more recent estimates from suggest interactions and mortality has decreased from pre-regulatory periods, with a conservative estimate of 26,500 loggerheads captured annually in US Atlantic fisheries causing mortality to 1,400 individuals per year (Finkbeiner *et al.* 2011). Pacific bycatch is much less, with about 400 individuals bycaught annually in US fisheries resulting in at least 20 mortalities (Finkbeiner *et al.* 2011). Each year, various fisheries capture about 2,000 loggerhead sea turtles in Pamlico Sound, of which almost 700 die. As a result of the 2006 and 2007 tri-national fishermen's exchanges in 2007 a prominent Baja California Sur fleet retired its bottom-set longlines (Peckham *et al.* 2008) (Peckham and Maldonado-Diaz 2011). Prior to this closure, the longline fleet interacted with an estimated 1,160-2,174 loggerheads annually, with nearly all (89 percent) of the takes resulting in mortalities(Peckham *et al.* 2008). Offshore longline tuna and swordfish longline fisheries are also a serious concern for the survival and recovery of loggerhead sea turtles and appear to affect the largest individuals more than younger age classes (Bolten *et al.* 1994, Aguilar *et al.* 1995, Howell *et al.* 2008, Tomás *et al.* 2008, Carruthers *et al.* 2009, Marshall *et al.* 2009, Petersen *et al.* 2009). Deliberate hunting of loggerheads for their meat, shells, and eggs has declined from previous exploitation levels, but still exists and hampers recovery efforts (Lino *et al.* 2010). In the Pacific, loggerhead turtles are captured, injured, or killed in numerous Pacific fisheries including:

- Japanese longline fisheries in the western Pacific Ocean and South China Seas
- direct harvest and commercial fisheries off Baja California, Mexico
- commercial and artisanal swordfish fisheries off Chile, Columbia, Ecuador, and Peru
- purse seine fisheries for tuna in the eastern tropical Pacific Ocean
- California/Oregon drift gillnet fisheries (NMFS 2006e)

Wallace *et al.* (2010) estimated that between 1990 and 2008, at least 85,000 sea turtles were captured as bycatch in fisheries worldwide. This estimate is likely at least two orders of magnitude low, resulting in a likely bycatch of nearly half a million sea turtles annually (Wallace *et al.* 2010); many of these are expected to be loggerhead sea turtles.

Marine debris ingestion can be a widespread issue for loggerhead sea turtles. More than one-third of loggerheads found stranded or bycaught had injected marine debris in a Mediterranean study, with possible mortality resulting in some cases (Lazar and Gračan 2010).

Climate change may also have significant implications on loggerhead populations worldwide. In addition to potential loss of nesting habitat due to sea level rise, loggerhead sea turtles are very sensitive to temperature as a determinant of sex while incubating. Ambient temperature increase by just 1º-2º C can potentially change hatchling sex ratios to all or nearly all female in tropical and subtropical areas (Hawkes *et al.* 2007a). Over time, this can reduce genetic diversity, or even population viability, if males become a small proportion of populations (Hulin *et al.* 2009). Sea surface temperatures on loggerhead foraging grounds correlate to the timing of nesting, with higher temperatures leading to earlier nesting (Mazaris *et al.* 2009a, Schofield *et al.* 2009). Increasing ocean temperatures may also lead to reduced primary productivity and eventual food availability. This has been proposed as partial support for reduced nesting abundance for loggerhead sea turtles in Japan; a finding that could have broader implications for other populations in the future if individuals do not shift feeding habitat (Chaloupka *et al.* 2008b). Warmer temperatures may also decrease the energy needs of a developing embryo (Reid *et al.* 2009).

Tissues taken from loggerheads sometimes contain very high levels of organochlorines chlorobiphenyl, chlordanes, lindane, endrin, endosulfan, dieldrin, PFOS, PFOA, DDT, and PCB (Rybitski *et al.* 1995, Mckenzie *et al.* 1999, Corsolini *et al.* 2000, Gardner *et al.* 2003, Keller *et al.* 2004a, Keller *et al.* 2004b, Keller *et al.* 2005, Alava *et al.* 2006, Perugini *et al.* 2006, Storelli *et al.* 2007a, Monagas *et al.* 2008, Oros *et al.* 2009). It appears that levels of organochlorines have the potential to suppress the immune system of loggerhead sea turtles and may affect metabolic regulation (Keller *et al.* 2004c, Keller *et al.* 2006, Oros *et al.* 2009). These contaminants could cause deficiencies in endocrine, developmental, and reproductive health (Storelli *et al.* 2007a). It is likely that the omnivorous nature of loggerheads makes them more prone to bioaccumulating toxins than other sea turtle species (Godley *et al.* 1999, Mckenzie *et al.* 1999).

Heavy metals, including arsenic, barium, cadmium, chromium, iron, lead, nickel, selenium, silver, copper, zinc, and manganese, have also been found in a variety of tissues in levels that increase with turtle size (Godley *et al.* 1999, Saeki *et al.* 2000, Anan *et al.* 2001, Fujihara *et al.* 2003, Gardner *et al.* 2006, Storelli *et al.* 2008, Garcia-Fernandez *et al.* 2009). These metals likely originate from plants and seem to have high transfer coefficients (Anan *et al.* 2001, Celik *et al.* 2006, Talavera-Saenz *et al.* 2007).

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

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

Critical habitat. The NMFS has not designated critical habitat for loggerhead sea turtles.

Anadromous fish

Gulf of Maine Atlantic salmon

Description of the species. Gulf of Maine (GOM) DPS Atlantic salmon occur along the Atlantic coast from the Androscoggin River (Maine) in the south to the St. Croix River on the US-Canadian border. The lower Penobscot River has three primary tributaries that contain Atlantic salmon: Cove Brook, Kenduskeag Stream, Kennebec and Ducktrap rivers. The estimated population of Atlantic salmon in the lower Penobscot River and its tributaries is less than 20 adult Atlantic salmon. Atlantic salmon are also listed in the Dennys River, East Machias River, Machias River, Pleasant River, Narraguagus River, and Sheepscot River.

Distribution. The Atlantic salmon is an anadromous fish species that is native to the basin of the North Atlantic Ocean from the Arctic Circle to Portugal in the eastern Atlantic Ocean, from Iceland and southern Greenland, and from the Ungava region of northern Quebec south to the Connecticut River (Scott and Crossman 1973a). In the US, Atlantic salmon historically ranged from Maine south to Long Island Sound. However, the central New England and Long Island Sound DPSs have been extirpated (65 FR 69459).

Habitat. The salmon's preferred spawning habitat is coarse gravel or rubble substrate (up to 3.5 inches in diameter) with adequate water circulation to keep the buried eggs well oxygenated (Peterson 1978). Water depth at spawning sites is typically between one and 2 feet deep, and water velocity averages 2 feet per second (Beland 1984). Spawning sites, or redds, average 8 feet long and 4.5 feet wide and are often located at the downstream end of riffles where water percolates through the gravel or where upwellings of groundwater occur (Moir *et al.* 1998). The annual egg production is approximately 240 eggs per 1,075 feet^2 of fluvial habitat (Chaput *et al.* 1998).

Movement, growth, and reproduction. Adult Atlantic salmon ascend the rivers of New England beginning in the spring and continuing into the fall, with peak numbers occurring in June. Although spawning does not occur until late fall, the majority of Atlantic salmon in Maine enter freshwater between May and mid-July (Meister 1958, Baum 1997). Salmon that return in early spring spend nearly 5 months in the river before spawning, often seeking cool water refuge (e.g., deep pools, springs, and mouths of smaller tributaries) during the summer months. Once an adult salmon enters a river, rising river temperatures and water flows stimulate upstream migration. Approximately 80% of salmon return to their home river after two years at sea, measuring approximately 2.5 feet long and weighing approximately 10 pounds (USFWS 2005b). A minority (10 to 20%) of Maine salmon return as smaller fish, or grilse, after only one winter at sea and still fewer return after three years at sea. A spawning run of salmon with representation of several age groups ensures some level of genetic exchange among generations. Once in freshwater, adult salmon cease feeding during their up-river migration. Spawning occurs in late October through November. Spawning sites are positioned within flowing water, particularly where upwelling of groundwater occurs, allowing for percolation of water through the gravel (Danie *et al.* 1984). These sites are most often positioned at the head of a riffle (Beland *et al.* 1982), the tail of a pool; or the upstream edge of a gravel bar where water depth is decreasing, water velocity is increasing (White 1942, McLaughlin and Knight 1987), and hydraulic head allows for permeation of water through the redd (a gravel depression where eggs are deposited).

A single female may create several redds before depositing all of her eggs. Female anadromous Atlantic salmon produce a total of 1,500-1,800 eggs per kilogram of body weight, yielding an average of 7,500 eggs per two sea-winter (SW) female (an adult female that has spent two winters at sea before returning to spawn) (Baum and Meister 1971).

After spawning, most Atlantic salmon move immediately downstream to backwater habitats near the head of tide (Cunjak *et al.* 1998, Fay *et al.* 2006). Upon returning to salt water, the spawned salmon or kelt resume feeding. If the salmon survives another one or two years at sea, it will return to its home river as a repeat spawner. From 1967 to 2003, approximately 3% of the wild and naturally reared adults that returned to rivers where adult returns are monitored--mainly the Penobscot River--were repeat spawners (USASAC 2004). Hatchery fish also return to the rivers into which they are stocked (Gorsky *et al.* 2009).

In late March or April, the eggs hatch into alevins. Alevins remain in the redd for about six weeks and are nourished by their yolk sac. Alevins emerge from the gravel about mid-May, generally at night, and begin actively feeding (Gustafson-Greenwood and Moring 1991). Survival from the egg to fry stage in Maine is estimated to range from 15-35% (Jordan and Beland 1981). Survival rates of eggs and larvae are a function of stream gradient, overwinter temperatures, interstitial flow, predation, disease, and competition (Bley and Moring 1988b). Once larval fry emerge from the gravel and begin active feeding they are referred to as fry. The majority of fry (>95%) emerge from redds at night (Gustafson-Marjenan and Dowse 1983). The survival rate of fry is affected by stream gradient, overwintering temperatures and water flows, and the level of predation and competition (Bley and Moring 1988a).

Within days, the free-swimming fry enter the parr stage, moving downstream to areas with adequate cover (rocks, vegetation, overhanging banks, and woody debris), water depths ranging from approximately four to 24 inches, velocities between 1foot and 3 feet per second, and temperatures near 61ºF (Beland 1984). When they finally reach their desired habitats, parr will actively defend territories that vary in size depending on the amount of food available and the density of other parr in the area (Symons 1971, McCormick *et al.* 1998, Armstrong *et al.* 1999). Some male parr become sexually mature and can successfully spawn with sea-run adult females. Water temperature, appetite, parr density, photoperiod, the level of competition and predation, and food supply may all influence the growth rate of parr (Lundqvist 1980, Randall 1982, Hearn 1987, Fausch 1988, Metcalfe *et al.* 1988, Elliot 1991, Nicieza and Metcalfe 1997). Maine Atlantic salmon parr densities are typically between three and nine parr per 1,075 feet^2, with years up to 16 parr per 1,075 feet^2 not uncommon (Beland 1996). There is no evidence of density-dependent limitations at densities of 13 parr per 1,075 feet^2 (Whalen *et al.* 2000). Parr feed on larvae of mayflies, stoneflies, caddisflies, chironomids, blackflies, annelids, and mollusks, as well as numerous terrestrial insects that fall into the river (Scott and Crossman 1973a).

In a parr's second or third spring, when it has grown 5 to 6 inches long, physiological, morphological, and behavioral changes occur (Schaffer and Elson 1975). This process, called smoltification, prepares parr for the dramatic change in osmoregulatory needs that comes with the transition from a freshwater to a saltwater habitat (Hoar 1976, McLeese *et al.* 1994, McCormick *et al.* 1998). In southern latitudes, including New England, most parr smolt after one year, but in cooler areas, they may take two to four years in freshwater before smolting (McCormick *et al.* 1998). Most smolts in New England rivers enter the sea during May and June to begin their ocean migration. Maine rivers produce approximately three smolts per 1,075 feet^2 of habitat.

Atlantic salmon of US origin are highly migratory, undertaking long marine migrations from the mouths of US rivers into the northwest Atlantic Ocean, where they are distributed seasonally over much of the region (Reddin 1985). The marine phase starts with smoltification and subsequent migration through the natal river and estuary. Upon completion of the physiological transition to saltwater, the post-smolt stage grows rapidly and has been documented moving in small, loosely aggregated schools near the surface (Dutil and Coutu 1988). After entering the nearshore waters of Canada, the post-smolts become part of a mixture of stocks of Atlantic salmon from various North American streams. Post-smolts appear to feed opportunistically on macroinvertebrates, amphipods, euphausiids, and fish (Hansen and Pethon 1985, Hansen and Quinn 1998, Andreassen *et al.* 2001). Once they mature to adult salmon, they travel individually and primarily eat capelin, herring, and sand lance (Hansen and Pethon 1985, Reddin 1985, Hansen and Quinn 1998).

Status and trends. The GOM DPS of anadromous Atlantic salmon was listed by the USFWS and NMFS as an endangered species on November 17, 2000 (65 FR 69495). The GOM DPS encompasses all naturally reproducing remnant populations of Atlantic salmon downstream of the former Edwards Dam site on the Kennebec River northward to the mouth of the St. Croix River. To date, Atlantic salmon are listed in the Dennys, East Maccias, Machias, Pleasant, Narraguagus, Ducktrap, and Sheepscot Rivers, Kenduskeag Stream, and Cove Brook. Naturally reproducing Atlantic salmon in the Penobscot River and its tributaries downstream of the former Bangor Dam are listed as endangered. The USFWS's GOM DPS river-specific hatchery-reared fish are also included as part of the listed entity (73 FR 51415).

Anadromous Atlantic salmon were native to nearly every major coastal river north of the Hudson River, New York (USFWS 2005b). The annual historic Atlantic salmon adult population returning to US rivers has been estimated to be between 300,000 and 500,000 (Stolte 1981, Beland 1984). The largest historical salmon runs in New England were likely in the Connecticut, Merrimack, Androscoggin, Kennebec, and Penobscot Rivers.

By the early 1800s, Atlantic salmon runs in New England had been severely depleted, reducing the distribution in the southern half of its range. Restoration efforts were initiated in the mid-1800s, but there was little success (Stolte 1981). There was a brief period of success in the late 19th century when limited runs were reestablished in the Merrimack and Connecticut Rivers by artificial propagation, but these runs were extirpated by the end of the century. By the end of the 19th century, three of the five largest salmon populations in New England (Connecticut, Merrimack, and Androscoggin Rivers) had been eliminated.

Abundance of adult Atlantic salmon is estimated using traps at a fishway, or through redd (nest) counts. Total trap counts, which include wild and hatchery fish, and total number of redds counted in GOM DPS between 1997 and 2004 are depicted in Figure 5. Such counts typically underestimate the actual returns of Atlantic salmon, but can give an idea of trends over time for index reaches and watershed. Juvenile smolt production is another measure of population trends, growth rate, and densities.

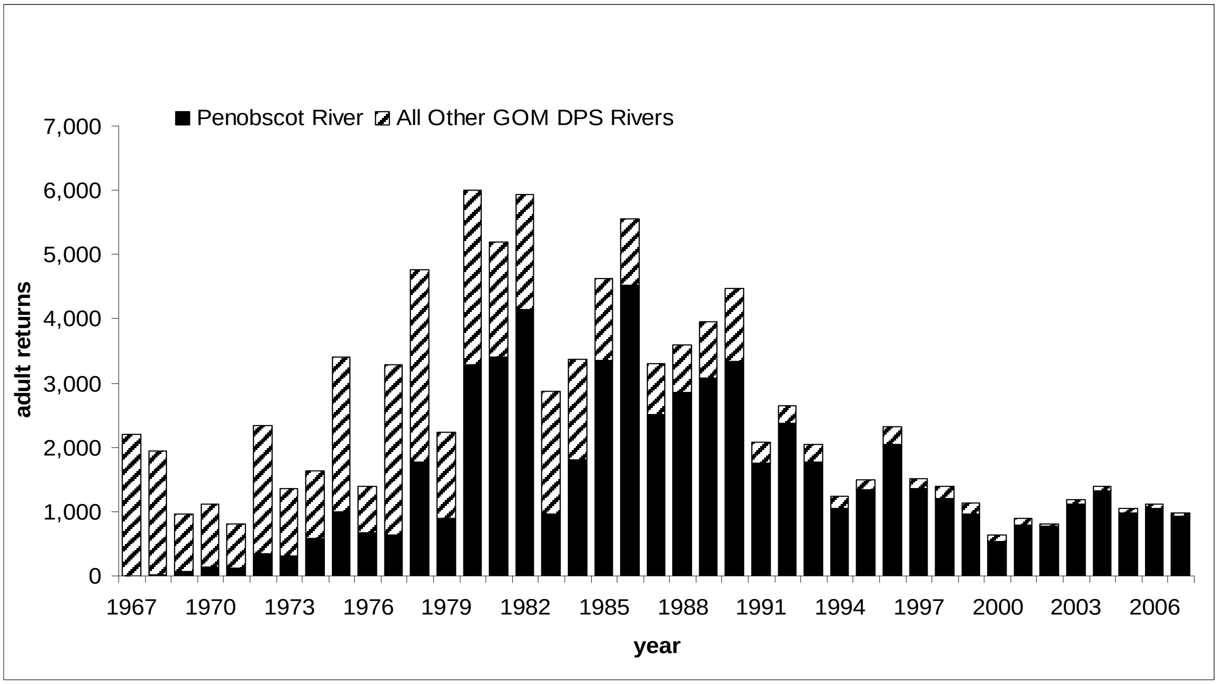

Figure 5. Adult returns to the GOM DPS 1967-2007.

Recently, Fay *et al.* (2006) used Population Viability Analysis (PVA) techniques to determine the conservation status of Atlantic salmon in the GOM DPS. Composite spawner data used to populate the model included adult return and rod kill estimates from the Penobscot River, adult spawner and rod kill estimates for the Narraguagus River, and adult spawner estimates for the GOM DPS. Using two time series, 1984 to 2004 and 1991 to 2004, Fay *et al.* (2006) calculated the negative population growth rates (for 1980-2004, lambda = 0.9690, variance = 0.0261; for 1991-2004, lambda = 0.9471, variance = 0.0142). From this, the estimated risk of extinction (defined herein as the number of spawners that falls below 100 individuals) within 100 years is 61% and 75% (or 28% and 45% in 40 years), for each respective data set.

Natural threats. Geographic features, such as waterfalls, pose natural barriers to salmon migration to spawning habitat. A variety of diseases affect Atlantic salmon, but are exacerbated by the presence of farming pens near river mouths. Atlantic salmon are prey for a variety of predators, including seals, porpoises, dolphins, otters, minks, birds, sharks, and a variety of other fishes at various salmon life stages.

Anthropogenic threats. Humans pose numerous threats to Atlantic salmon survival and recovery (see USFWS 2005b for a review). Water quality in both marine, estuarine, and aquatic habitats suffers from both point and non-point source pollution, both biological (bacteria) and chemical. Riverine environments are becoming acidified, which can cause physiological stress in adults and altered developmental biology in eggs or hatchlings. In association with acidification, aluminum toxicity can lead to osmoregulation failure. This is because Atlantic salmon are highly sensitive to pH changes and many runs of Atlantic salmon in Sweden, Norway, and Canada have been severely depleted or extirpated due to acidity changes in river systems resulting from industrial activity (Watt 1981, Watt *et al.* 1983, Watt *et al.* 2000, Sandøy and Langåker 2001). Pesticide use and its immigration into Maine waterways is also of concern.

For example, atrazine can significantly impair water balance in salmon even at low concentrations, resulting in a reduced ability for salmon to move between fresh and salt water (Jagoe and Haines 1990, Staurnes *et al.* 1993, WWF 2001). At levels that presently occur in stream environments, male salmon also experience impaired olfactory reception in being able to detect female pheromones (Waring and Moore 1998). Thus, male reproduction activity is not cued to that of females and has the potential to severely reduce recruitment. Nonylphenols are also severely detrimental to juvenile salmon. These chemicals also reduce the ability of smolts to transition between fresh and salt water, leading to mortality, as well as imitate female hormones leading to eggs that do not hatch (Fairchild *et al.* 1999, WWF 2001). Sedimentation due to erosion and development in and around aquatic waterways can degrade salmon habitat and the habitat of their invertebrate prey. Excessive nutrient load, as in marine systems, can lead to a bloom of plant growth and subsequent death, which reduces oxygen levels to anoxic conditions. This can lead to extensive habitat loss and salmon mortality.

Although changes overtly seem minor, increases in Maine's river temperatures can have broad impacts on salmon recovery, including changes in fish physiology, prey abundance and distribution, loss of spawning activity, and other effects (USFWS 2005b, Holbrook *et al.* 2009). As in Pacific salmon species, Atlantic salmon decline originated largely from manmade barriers across rivers preventing movement to and from spawning and marine habitats. Although many of these barriers have since been modified or removed, modern construction (bridges, culverts, etc.) that do not consider Atlantic salmon needs can hinder recovery efforts (Holbrook *et al.* 2009). When water temperatures exceed 22º C during spawning runs, Atlantic salmon tend to have poorer success in passing obstacles than (Holbrook *et al.* 2009).

Atlantic salmon fisheries have been discontinued in the US, Canada, and Greenland. A high threat is posed by farm-raised salmon due to the potential for these fish to escape (instances of thousands of fish escaping are known) and interbreed with wild salmon, thereby affecting the genetics of Atlantic salmon as a species. Recent evidence shows that supportive breeding programs, where wild Atlantic salmon are captured and bred in captivity and young are released early in life, produce fish that are genetically, morphologically, and behaviorally different from truly wild progeny (Blanchet *et al.* 2008). The presence of disease and parasites in farm-raised salmon pens can also have a deleterious effect on wild Atlantic salmon.

Climate change has the potential to be a strong negative influence on Altantic salmon. Remaining occupied habitat is at the southern edge of the ESU's range. To survive, populations have adapted to distinct physical and environmental conditions here (Saunders 1981). Climate models predict significant, extended warming (IPCC 2001b). Although periods of North Atlantic warming and cooling have occurred, changes have not been uniform as global warming is, changing sea temperatures, wind currents, fresh water input, and mixing of the ocean's surface layer. Small thermal changes can critically affect biological functions, such as protein metabolism, response to aquatic contaminants, reproductive performance, smolt development, and species distribution limits (Keleher and Rahel 1996, McCormick *et al.* 1997, Reid *et al.* 1997, Somero and Hofmann 1997, Van der Kraak and Pankhurst 1997, Welch *et al.* 1998). Atlantic salmon smolt growth is known to change with temperature, with a temperature increase from 57º to 64ºF resulting in a greater than 10% decrease in growth rate (Handeland *et al.* 2008).

It should be noted that positive effects may also be realized by climate change and specifically warmer water temperature. Increased opportunities for growth in spring and summer could increase the percentage of fish that enter the upper size distribution of a population and smolt the following spring (Thorpe 1977, Thorpe *et al.* 1980, Thorpe 1994). In addition, warmer rearing temperatures during the late winter and spring have been shown to advance the timing of the parr-smolt transformation in Atlantic salmon (Solbakken *et al.* 1994). There is, however, an optimal temperature range and a limit for growth after which salmon parr will stop feeding due to thermal stress. During this time, protein degradation and weight loss will increase with rising water temperature (McCarthy and Houlihan 1997). The NRC (2004) concluded that some degree of climate warming or change in hydrologic regime could be tolerated if other problems affecting Atlantic salmon are reduced.

Critical habitat. On June 19, 2009, 45 specific areas occupied by Atlantic salmon at the time of listing (approximately 19,571 km of perennial river, stream, and estuary habitat and 799 square kilometers of lake habitat within the range of the GOM DPS) were established for Atlantic salmon critical habitat (74 FR 29300). The PCEs for this critical habitat include:

- Deep, oxygenated pools and cover (e.g., boulders, woody debris, vegetation, etc.), near freshwater spawning sites, necessary to support adult migrants during the summer while they await spawning in the fall.
- Freshwater spawning sites that contain clean, permeable gravel and cobble substrate with oxygenated water and cool water temperatures to support spawning activity, egg incubation, and larval development as well as support emergence, territorial development and feeding activities of Atlantic salmon fry.
- Freshwater rearing sites with space to accommodate growth and survival of Atlantic salmon parr.
- Freshwater rearing sites with a combination of river, stream, and lake habitats that accommodate parr's ability to occupy many niches and maximize parr production.
- Freshwater rearing sites with cool, oxygenated (6 mg/L) water and diverse food resources (mayflies, stoneflies, chironomids, caddisflies, blackflies, aquatic annelids, and mollusks, as well as numerous terrestrial invertebrates, alewives, dace, or minnows) to support growth and survival of Atlantic salmon parr.
- Freshwater and estuary migratory sites free from physical and biological barriers that delay or prevent access of adult salmon seeking spawning grounds needed to support recovered populations or prevent emigration of smolts to the marine environment.
- Freshwater and estuary migration sites with pool, lake, and instream habitat that provide cool, oxygenated water and cover items (e.g., boulders, woody debris, and vegetation) to serve as temporary holding and resting areas during upstream migration of adult salmon.
- Freshwater and estuary migration sites with abundant, diverse native fish communities to serve as a protective buffer against predation.
- Freshwater and estuary migration sites with sufficiently cool water temperatures and water flows that coincide with diurnal cues to stimulate smolt migration.
- Freshwater migration sites with water chemistry (particularly pH) needed to support sea water adaptation of smolts.

These PCEs have undergone significant degradation over in the recent past. Dams, along with degraded substrate and cover, water quality, water temperature, and biological communities, have reduced the quality and quantity of habitat available to Atlantic salmon populations within the DPS. A combined total of twenty FERC-licensed hydropower dams on the Penobscot River significantly impede the migration of Atlantic salmon and other diadromous fish to historically accessible spawning and rearing habitat. Agriculture and urban development largely affect the lower third of the Penobscot River below the Piscataquis River sub-basin by reducing substrate and cover, reducing water quality, and elevating water temperatures. Introductions of smallmouth bass and other non-indigenous species significantly degrade habitat quality throughout the mainstem Penobscot and portions of the Mattawamkeag, Piscataquis, and lower Penobscot sub-basins by altering predator/prey relationships. Similar to smallmouth bass, recent Northern pike introductions threaten habitat in the lower Penobscot River below the Great Works Dam.

Today, dams are the greatest impediment, outside of marine survival, to the recovery of salmon in the Penobscot, Kennebec and Androscoggin river basins (Fay *et al.* 2006). Hydropower dams significantly impede the migration of Atlantic salmon and other diadromous fish and either reduce or eliminate access to historically accessible spawning and rearing habitat. In addition to hydropower dams, agriculture and urban development largely affect the lower third of the Merrymeeting Bay recovery unit by reducing substrate and cover, reducing water quality, and elevating water temperatures. Additionally, smallmouth bass and brown trout introductions, along with other non-indigenous species, significantly degrade habitat quality throughout the Merrymeeting Bay recovery unit by altering natural predator/prey relationships.

Impacts to substrate and cover, water quality, water temperature, biological communities, and migratory corridors, among a host of other factors, have impacted the quality and quantity of habitat available to Atlantic salmon populations within the Downeast Coastal recovery unit. Two hydropower dams on the Union river, and to a lesser extent the small ice dam on the lower Narraguagus River, limit access to spawning and rearing habitat within these two watersheds. In the Union River, physical and biological features have been most notably limited by high water temperatures and abundant smallmouth bass populations associated with impoundments. In the Pleasant River and Tunk Stream, which collectively contain over 4,300 units of spawning and rearing habitat, pH has been identified as possibly being the predominate limiting factor. The Machias, Narraguagus, and East Machias rivers contain the highest quality habitat and collectively account for approximately 40 percent of the spawning and rearing habitat in the Downeast Coastal recovery unit.

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

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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

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 56º 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.

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 21.9º C (mean average body temperature), 20.6º C (mean minimum daily body temperature), or 23.1º 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.

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

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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.

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, 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, 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).

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).

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 2011l). 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 2011l).

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).

Sacramento River winter-run Chinook salmon

Distribution. The Sacramento River winter-run Chinook salmon ESU includes all naturally spawned populations of winter-run Chinook salmon in the Sacramento River and its tributaries in California. Four natural, independent populations once existed (NMFS 2011g). Two artificial propagation programs are included in this ESU: winter-run Chinook salmon from the Livingston Stone National Fish Hatchery, and winter-run Chinook salmon in a captive broodstock program maintained at the Livingston Stone National Fish Hatchery and the University of California Bodega Marine Laboratory. 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.

This ESU consists of a single spawning population that enters the Sacramento River and its tributaries in California from November to June and spawns from late April to mid-August, with a peak from May to June (Table 7). Sacramento River winter-run Chinook salmon historically occupied cold, headwater streams, such as the upper reaches of the Little Sacramento, McCloud, and lower Pit Rivers. Young winter-run Chinook salmon venture to sea in November and December, after only four to seven months in fresh water (Groot *et al.* 1991).

Table 7. Sacramento River winter-run Chinook salmon abundance and selected measures of population viability.

Population	Historical Abundance[a]	Mean number of Spawners (Range)[b]	Percent Hatchery Contribution	Population growth rate (λ)[c]
Sacramento River winter-run	200,000	2,191 (364-65,683)	<10	0.97 (0.87, 1.09)

[a] Historical abundance for the total ESU based on commercial fishery landings in the 1870s (Fisher 1994). Individual river estimates of historical abundance not provided.

[b] Recent geometric mean number of spawners from Good *et al.* 2005.

[c] Lambda value reported by Good *et al.* 2005. The 90% confidence intervals are noted in parentheses.

Status and trends. NMFS listed Sacramento River winter-run Chinook salmon as endangered on January 4, 1994 (59 FR 440), and reaffirmed their status as endangered on June 28, 2005 (70 FR 37160), because dams restrict access to a small fraction of their historic spawning habitat and the habitat remaining to them is degraded. Sacramento River winter-run Chinook salmon consist of a single self-sustaining population which is entirely dependent upon the provision of suitably cool water from Shasta Reservoir during periods of spawning, incubation and rearing.

Construction of Shasta Dams in the 1940s eliminated access to historic spawning habitat for winter-run Chinook salmon in the basin. Winter-run Chinook salmon were not expected to survive this habitat alteration (Moffett 1949). However, cold water releases from Shasta Dam have created conditions suitable for winter Chinook salmon for roughly 60 miles downstream from the dam. As a result the ESU has been reduced to a single spawning population confined to the mainstem Sacramento River below Keswick Dam, although some adult winter-run Chinook salmon were recently observed in Battle Creek, a tributary to the upper Sacramento River.

Quantitative estimates of run-size are not available for the period before 1996, the completion of Red Bluff Diversion Dam. However, winter-runs may have been as large as 200,000 fish based upon commercial fishery records from the 1870s (Fisher 1994). The California Department of Fish and Game estimated spawning escapement of Sacramento River winter-run Chinook salmon at 61,300 (60,000 in the mainstem, 1,000 in Battle Creek, and 300 in Mill Creek) in the early 1960s. During the first 3 years of operation of the county facility at the Red Bluff Diversion Dam (1967 to 1969), the spawning run of winter-run Chinook salmon averaged 86,500 fish. From 1967 through the mid-1990s, the population declined at an average rate of 18% per year, or roughly 50% per generation. The population reached critically low levels during the drought of 1987 to 1992; the 3-year average run size for the period of 1989 to 1991 was 388 fish. Based on the Red Bluff Diversion Dam counts, the population has been growing rapidly since the 1990s. Mean run size from 1995-2000 has been 2,191, but have ranged from 364 to 65,683 (Good *et al.* 2005b). Good *et al.* 2005 estimated that the short term trend is 0.26, while the population growth rate is still less than 1 (Table 13). The draft recovery goal for the ESU is an average of 10,000 female spawners per year and a population growth rate >1.0, calculated over 13 years of data (Good *et al.* 2005b). Since 2004, escapement in the central Valley has fallen dramatically from nearly 16,000 in 2004 to about 1,500 in 2010; abundance peaked in 2006 (NMFS 2011g). Abundance has fallen throughout the ESU since 2000 (NMFS 2011g).

Critical habitat. NMFS designated critical habitat for Sacramento River winter-run Chinook salmon on June 16, 1993 (58 FR 33212). The following areas consisting of the water, waterway bottom, and adjacent riparian zones: the Sacramento River from Keswick Dam, Shasta County (river mile 302) to Chipps Island (river mile 0) at the westward margin of the Sacramento-San Joaquin Delta, and other specified estuarine waters. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. Factors contributing to the downward trends in this ESU include reduced access to spawning/rearing habitat, possible loss of genetic integrity through population bottlenecks, inadequately screened diversions, predation at artificial structures and by nonnative species, pollution from Iron Mountain Mine and other sources, adverse flow conditions, high summer water temperatures, unsustainable harvest rates, passage problems at various structures, and vulnerability to drought (Good *et al.* 2005b).

California coastal Chinook salmon

Distribution. The California Coastal Chinook salmon ESU includes all naturally spawned populations of Chinook salmon from rivers and streams south of the Klamath River to the Russian River, Californian. Seven artificial propagation programs are part of this ESU: The Humboldt Fish Action Council (Freshwater Creek), Yager Creek, Redwood Creek, Hollow Tree, Van Arsdale Fish Station, Mattole Salmon Group, and Mad River Hatchery fall-run Chinook hatchery programs. 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.

California Coastal Chinook salmon are a fall-run, ocean-type fish. A spring-run (river-type) component existed historically, but is now considered extinct (Bjorkstedt *et al.* 2005). Table 8 identifies populations within the California Coastal Chinook salmon ESU, their abundances, and the relative contribution of artificially propagated fish to the population. Although difficult to identify, it appears that 17 populations comprise the ESU (NMFS 2011a).

Table 8. California coastal Chinook populations and selected measures of population viability.

Population	Historical Abundance[a]	Mean Number of Spawners (Range)[b]	Percent Hatchery Contribution[c]	Long-term Trend[d]
Freshwater Creek		22 (13-22)	30-70	0.137 (-0.405, 0.678)
Eel River	17,000-55,000		~30	
Mainstem Eel River	13,000			
Sprowl Creek		43 (43-497)		-0.096 (-0.157, -0.034)
Tomki Creek		61 (13-2233)		-0.199 (-0.351, -0.046)
Van Duzen River	2,500			
Middle Fork Eel River	13,000			
South Fork Eel River	27,000			
North Fork Eel River				
Upper Eel River				
Redwood Creek	1,000-5,000			
Mad River	1,000-5,000			
Canyon Creek		73 (19-103)		0.0102 (-0.106, 0.127)
Bear River	100			
Mattole River	1,000-5,000		~17	
Russian River	50-500		~0	
Humbolt Bay tributaries	40			
Tenmile to Gualala			0	
Small Humboldt County rivers	1,500		0	
Rivers north of Mattole River	600		0	
Noyo River	50		0	

[a] Historical abundance estimates based on professional opinion and evaluation of habitat conditions (reported in Good *et al.* 2005).

[b] 5-year (1997-2001) geometric mean number of counts of adults (quasi-systematic surveys of spawners – Canyon, Tomki, and Sprowl creeks; returning spawners at Freshwater Creek weir).

[c] Hatchery production in this ESU is at low levels, aimed at supplementing depressed runs. Operational procedures and low production suggest that the ESU may not be at substantial risk of degraded genetic integrity (Good *et al.* 2005).

[d] Long-term trends were calculated using the entire available data set (see Good *et al.* 2005). The 90% confidence intervals are noted in parentheses.

Status and trends. NMFS listed California Coastal Chinook salmon as threatened on September 16, 1999 (64 FR 50393), and they retained their threatened status on June 28, 2005 (70 FR 37160). California Coastal Chinook salmon were listed due to the combined effect of dams that prevent them from reaching spawning habitat, logging, agricultural activities, urbanization, and water withdrawals in the river drainages that support them. Historical estimates of escapement, based on professional opinion and evaluation of habitat conditions, suggest abundance was roughly 73,000 in the early 1960s with the majority of fish spawning in the Eel River (CDFG, 1965 #272 in Good *et al.* 2005b). The species exists as small populations with highly variable cohort sizes. Sproul Creek abundance (part of the lower Eel River population) has been on a positive trend since 1995, but remains negative over longer time series (NMFS 2011a). The Russian River probably contains some natural production, but the origin of those fish is not clear because of a number of introductions of hatchery fish over the last century. Average number of returning spawners in the Russian River has slowly declined since 2001 (NMFS 2011a). The Eel River contains a substantial fraction of the remaining Chinook salmon spawning habitat for this species. Since its original listing and status review, little new data are available or suitable for analyzing trends or estimating changes in this population's growth rate (Good *et al.* 2005b). Adult spawning returns in Prairie Creek (part of the Redwood Creek population) have been on the decline since they were first monitored in 1998 (NMFS 2011a).

Long-term trends in Freshwater Creek were positive until 2008 and 2009, when only two fish were caught in weirs (NMFS 2011a). In Canyon Creek, although abundance trends are only slightly different than zero, the trend is positive (Table 14). Long-term trends in Sprowl and Tomki creeks (tributaries of the Eel River), however, are negative. Good *et al.* (2005b) caution making inferences on the basin-wide status of these populations as they may be weak because the data likely include unquantified variability due to flow-related changes in spawners' use of mainstem and tributary habitats. Unfortunately, none of the available data is suitable for analyzing the long-term trends of the ESU or estimating the population growth rate.

Spring-run Chinook appear to be extirpated from the north-caostal and north mountain interior strata of the ESU, representing a considerable loss of diversity (NMFS 2011a). In addition, all populations south of the Mattole River to the Russian River has resulted in the entire north-central coastal stratum being unoccupied and the central coastal stratum has only one declining population (NMFS 2011a).

Critical habitat. NMFS designated critical habitat for California Coastal Chinook salmon on September 2, 2005 (70 FR 52488). Specific geographic areas designated include the following CALWATER hydrological units: Redwood Creek, Trinidad, Mad River, Eureka Plain, Eel River, Cape Mendocino, Mendocino Coast, and the Russian River. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. The critical habitat designation for this ESU identifies primary constituent elements that include sites necessary to support one or more Chinook salmon life stages. Specific sites 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. The critical habitat designation (70 FR 52488) contains additional details on the sub-areas that are included as part of this designation, and the areas that were excluded from designation.

In total, California Coastal Chinook salmon occupy 45 watersheds (freshwater and estuarine). The total area of habitat designated as critical includes about 1,500 miles of stream habitat and about 25 square miles of estuarine habitat, mostly within Humboldt Bay. 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 bankfull elevation. In estuarine areas the lateral extent is defined by the extreme high water because extreme high tide areas encompass those areas typically inundated by water and regularly occupied by juvenile salmon during the spring and summer, when they are migrating in the nearshore zone and relying on cover and refuge qualities provided by these habitats, and while they are foraging. Of the 45 watershed reviewed in NMFS' assessment of critical habitat for California Coastal Chinook salmon, eight watersheds received a low rating of conservation value, 10 received a medium rating, and 27 received a high rating of conservation value for the species.

Critical habitat in this ESU consists of limited quantity and quality summer and winter rearing habitat, as well as marginal spawning habitat. Compared to historical conditions, there are fewer pools, limited cover, and reduced habitat complexity. The limited instream cover that does exist is provided mainly by large cobble and overhanging vegetation. Instream large woody debris, needed for foraging sites, cover, and velocity refuges is especially lacking in most of the streams throughout the basin. NMFS has determined that these degraded habitat conditions are, in part, the result of many human-induced factors affecting critical habitat including dam construction, agricultural and mining activities, urbanization, stream channelization, water diversion, and logging, among others.

Lower Columbia River Chinook salmon

Distribution. The Lower Columbia River Chinook salmon ESU includes all naturally spawned populations of Chinook salmon from the Columbia River and its tributaries from its mouth at the Pacific Ocean upstream to a transitional point between Washington and Oregon, east of the Hood River and the White Salmon River, and includes the Willamette River to Willamette Falls, Oregon, exclusive of spring-run Chinook salmon in the Clackamas River (Figure 6). Seventeen artificial propagation programs are part of this 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.

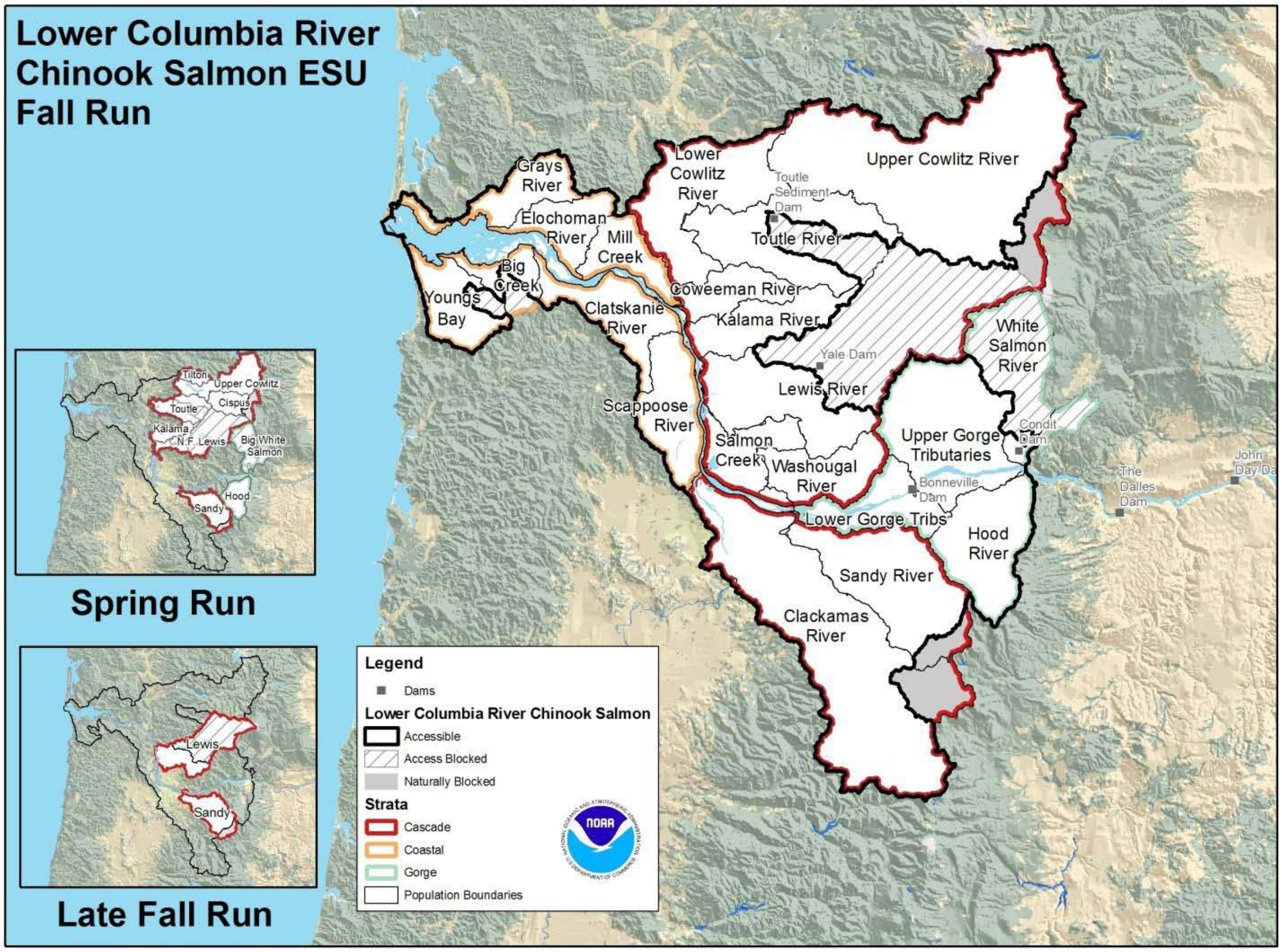

Figure 6. Population boundaries, dams, accessible areas, and extirpated reaches of lower Columbia River Chinook distribution, incorporating critical habitat boundaries (dark black line) and fall versus spriung runs. Taken from NMFS (2011).

Lower Columbia River Chinook salmon have three life history types: early fall runs (tules), late fall runs (brights), and spring-runs. Spring and fall runs have been designated as part of a Lower Columbia River Chinook salmon ESU. The Cowlitz, Kalama, Lewis, White Salmon, and Klickitat Rivers are the major river systems on the Washington side, and the lower Willamette and Sandy Rivers are foremost on the Oregon side. The eastern boundary for this species occurs at Celilo Falls, which corresponds to the edge of the drier Columbia Basin Ecosystem and historically may have been a barrier to salmon migration at certain times of the year. The predominant life history type for this species is the fall-run. Fall Chinook typically enter the Columbia River in August through October to spawn in the mainstems of large rivers (Kostow 1995). Spring Chinook enter freshwater in March through June to spawn in upstream tributaries and generally emigrate from freshwater as yearlings.

Status and trends. NMFS originally listed Lower Columbia River Chinook salmon as threatened on March 24, 1999 (64 FR 14308); NMFS reaffirmed the threatened status of Lower Columbia River Chinook salmon on June 28, 2005 (70 FR 37160). Of 32 historical populations, 30 remain and 28 are considered at high risk of extinction (NMFS 2011d). Spring-fun individuals are at particularly high risk, as they have been cut off from extensive spawning areas by dams (NMFS 2011d). Historical records of Chinook salmon abundance are sparse, but cannery records suggest a peak run of 4.6 million fish (43 million pounds) in 1883 (Lichatowich 1999). Although fall-run Chinook salmon are still present throughout much of their historical range, they are still subject to large-scale hatchery production, relatively high harvest, and extensive habitat degradation. The Lewis River late-fall-run population is the healthiest and has a reasonable probability of being self-sustaining. Abundances largely declined during 1998-2000 and trend indicators for most populations are negative, especially if hatchery fish are assumed to have a reproductive success equivalent to that of natural-origin fish (see Table 9).

Table 9. Lower Columbia River Chinook salmon life histories, populations and selected measures of population viability.

Life history	Population	Historical abundance[a]	Mean number of spawners (range)[b]	Percent hatchery contribution[c]	Long-term median growth rate (λ)[d]
Fall run	Youngs Bay				
	Grays River	2,477	99	38	0.944, 0.844
	Big Creek				
	Elochoman River		676	68	1.037, 0.800
	Clatskanie River[e]		50 (34-74)		0.99
	Mill, Abernathy, and Germany Creeks		734	47	0.981, 0.829
	Scappoose Creek				
	Coweeman River	4,971	274	0	1.092, 1.091
	Lower Cowlitz River	53,956	1,562	62	0.998, 0.682
	Upper Cowlitz River		5,682		
	Toutle River	25,.392			
	Kalama River	22,455	2,931	67	0.973, 0.818
	Salmon Creek and Lewis River	47,591[f]	256	0	0.984, 0.979
	Clackamas River		40		
	Washougal River	7,518	3,254	58	1.025, 0.815
	Sandy River		183		
	Columbia Gorge-lower tributaries				
	Columbia Gorge-upper tributaries	2,363	136 (Wind River only)	13 (Wind River only	0.959, 0.955
	Hood River		18		
	Big White Salmon River		334	21	0.963, 0.945
Late fall (bright)	Sandy River[e]		3085 (2337-4074)	3	0.997
	North Fork Lewis River		7,841	13	0.968, 0.948
Spring run	Upper Cowlitz River				
	Cispus River		1,787		
	Tilton River				
	Toutle River	2,901			
	Kalama River	4,178	98		
	Lewis River		347		
	Sandy River[e]		297 (202-438)		0.961
	Big White Salmon River				
	Hood River		51		

[a] Historical abundance for various rivers was calculated using the Ecosystem and Diagnosis Treatment (EDT) model, which attempts to predict population performance based on reach-specific habitat attributes. Estimates are provided

as a means of comparing the historical abundance of populations relative to current abundance. See Good *et al.* (2005) for a discussion about the uncertainty associated with these estimates.
[b] Recent geometric mean number of spawners as reported in Good *et al.* 2005
[c] Recent average hatchery-origin spawners (%) as reported by Good *et al.* 2005. Natural-origin spawners are those that had parents that spawned in the wild, as opposed to hatchery-origin fish, whose parents were spawned in a hatchery.
[d] The long-term median growth rate (λ) is an estimate of the natural growth rate after accounting for hatchery-origin spawners. The two values are estimates under two hypotheses about the reproductive success of hatchery origin spawners. Hatchery fish are assumed to have zero reproductive success in the first estimate. In the second estimate hatchery fish are assumed to have the same relative reproductive success as natural-origin fish. Growth rates were not calculated for all populations, as adequate data were not available (see Good *et al.* 2005 for 95% confidence intervals on growth estimates).
[d] Values for these populations are reported in McElhany *et al.* 2007, and represent estimates based on the total available data series, which varies by population.
[fe] Combined estimate of Lewis River fall run (East Fork only) and Lewis River brights (Good *et al.* 2005)

Most populations for which data are available have a long-term declining population trend (Table 15). Currently, the spatial extent of populations in the Coastal and Cascade fall runs are similar to their respective historical conditions. New data include spawner abundance estimates through 2001, new estimates of the fraction of hatchery spawners, and harvest estimates. In addition, estimates of historical abundance have been provided by the Washington Department of Fish and Wildlife. The Willamette/Lower Columbia River Technical Review Team estimated that 8-10 historic populations have been extirpated, most of them spring-run populations. Near loss of that important life history type remains an important concern. Although some natural production currently occurs in 20 or so populations, only one exceeds 1,000 spawners. Almost all spring-runs are at very high risk of extinction. High hatchery production continues to pose genetic and ecological risks to natural populations and to mask their performance for Coastal, Cascade, and Gorge fall run populations. Most Lower Columbia River populations have not seen increases in recent years as pronounced as those that have occurred in many other geographic areas. Although populations increased through most of the past decade, recent declines leave abundance near where it was in 2001 (NMFS 2011d).

Critical habitat. NMFS designated critical habitat for Lower Columbia River Chinook salmon on September 2, 2005 (70 FR 52630). Designated critical habitat includes all Columbia River estuarine areas and river reaches proceeding upstream to the confluence with the Hood River as well as specific stream reaches in a number of tributary subbasins. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. The critical habitat designation for this ESU identifies primary constituent elements that include sites necessary to support one or more life stages. Specific sites 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 52 subbasins reviewed in NMFS' assessment of critical habitat for the Lower Columbia River Chinook salmon ESU, 13 subbasins were rated as having a medium conservation value, four were rated as low, and the remaining subbasins (35), were rated as having a high conservation value to the ESU. Factors contributing to the downward trends in this ESU are hydromorphological changes resulting from hydropower development, loss of tidal marsh and swamp habitat, and degraded freshwater and marine habitat from industrial harbor and port development, and urban development. Limiting factors identified for this species include reduced access to spawning/rearing habitat in tributaries, hatchery impacts, loss of habitat diversity and channel stability in tributaries, excessive fine sediment in spawning gravels, elevated water temperature in tributaries, and harvest impacts.

On the mainstem of the Columbia River, hydropower projects, including the Federal Columbia River Hydropower System (FCRPS), have significantly degraded salmon and steelhead habitats (Bottom *et al.* 2005, Fresh *et al.* 2005, NMFS 2006d). The series of dams and reservoirs that make up the FCRPS block an estimated 12 million cubic yards of debris and sediment that would otherwise naturally flow down the Columbia River and replenish shorelines along the Washington and Oregon coasts.

Industrial harbor and port development are also significant influences on the Lower Willamette and Lower Columbia rivers (Bottom *et al.* 2005, Fresh *et al.* 2005, NMFS 2006d, LCFRB 2010). Since 1878, 100 miles of river channel within the mainstem Columbia River, its estuary, and Oregon's Willamette River have been dredged as a navigation channel by the ACOE. Originally dredged to a 20-foot minimum depth, the Federal navigation channel of the Lower Columbia River is now maintained at a depth of 43 feet and a width of 600 feet. The Lower Columbia River supports five ports on the Washington State side: Kalama, Longview, Skamania County, Woodland, and Vancouver. In addition to loss of riparian habitat, and disruption of benthic habitat due to dredging, high levels of several sediment chemicals, such as arsenic and polycyclic aromatic hydrocarbons (PAHs), have been identified in Lower Columbia River watersheds in the vicinity of the ports and associated industrial facilities.

The most extensive urban development in the Lower Columbia River subbasin has occurred in the Portland/Vancouver area. Outside of this major urban area, the majority of residences and businesses rely on septic systems. Common water quality issues with urban development and residential septic systems include higher water temperatures, lowered dissolved oxygen, increased fecal coliform bacteria, and increased chemicals associated with pesticides and urban runoff.

The Columbia River estuary has lost a significant amount of the tidal marsh and tidal swamp habitats that are critical to juvenile salmon and steelhead, particularly small or ocean-type species (Bottom *et al.* 2005, Fresh *et al.* 2005, NMFS 2006d, LCFRB 2010). Edges of marsh areas provide sheltered habitats for juvenile salmon and steelhead where food, in the form of amphipods or other small invertebrates which feed on marsh detritus, is plentiful, and larger predatory fish can be avoided. Historically, floodwaters of the Columbia River inundated the margins and floodplains along the estuary, allowing juvenile salmon and steelhead access to a wide expanse of low-velocity marshland and tidal channel habitats. In general, the riverbanks were gently sloping, with riparian and wetland vegetation at the higher elevations of the river floodplain becoming habitat for salmon and steelhead during flooding river discharges or flood tides. Sherwood *et al.* (1990) estimated that the Columbia River estuary lost 20,000 acres of tidal swamps, 10,000 acres of tidal marshes, and 3,000 acres of tidal flats between 1870 and 1970. This study further estimated an 80% reduction in emergent vegetation production and a 15% decline in benthic algal production.

Habitat and food-web changes within the estuary, and other factors affecting salmon population structure and life histories, have altered the estuary's capacity to support juvenile salmon

(Bottom *et al.* 2005, Fresh *et al.* 2005, NMFS 2006d, LCFRB 2010). Diking and filling activities have reduced the tidal prism and eliminate emergent and forested wetlands and floodplain habitats. These changes likely have reduced the estuary's salmon-rearing capacity. Moreover, water and sediment in the Lower Columbia River and its tributaries have toxic contaminants that are harmful to fish and wildlife (LCREP 2007). Contaminants of concern include dioxins and furans, heavy metals, polychlorinated biphenyls (PCBs) and organochlorine pesticides such as DDT. Simplification of the population structure and life-history diversity of salmon possibly is yet another important factor affecting juvenile salmon viability. Restoration of estuarine habitats, particularly diked emergent and forested wetlands, reduction of avian predation by terns, and flow manipulations to restore historical flow patterns may have begun to enhance the estuary's productive capacity for salmon, although historical changes in population structure and salmon life histories may prevent salmon from making full use of the productive capacity of estuarine habitats.

Species-specific threats and limitations to recovery. Lower Columbia River Chinook salmon face numerous hindrances to their survival and recovery, including degraded estuarine and near-shore marine habitat resulting from cumulative impacts of land use and flow management by the Columbia River hydropower system and floodplain connectivity and function, channel structure and complexity, riparian areas, stream substrate, stream flow, and water quality have been degraded as a result of cumulative impacts of agriculture, forestry, and development (LCFRB 2010, NMFS 2011d). Further habitat-based impacts to the species result from an altered flow regime and Columbia River plume has altered the temperature regime and estuarine food web, and has reduced ocean productivity, reduced access to off-channel rearing habitat in the lower Columbia River, and reduced productivity resulting from sediment and nutrient-related changes in the estuary. Also hampering the species is reduced access to spawning and rearing habitat mainly as a result of tributary hydropower projects, hatchery-related effects, and harvest-related effects on fall Chinook salmon, juvenile fish strandings that result from ship wakes, and contaminants affecting fish health and reproduction.

Central Valley spring-run Chinook salmon

Distribution. The Central Valley spring-run Chinook salmon ESU includes all naturally spawned populations of spring-run Chinook salmon in the Sacramento River and its tributaries in California. This ESU includes one artificial propagation program, the Feather River Hatchery spring-run Chinook salmon program. This artificially propagated population is no more divergent relative to the local natural populations than would be expected between closely related populations within this ESU. Only three extant populations remain from 18 or 19 original populations (NMFS 2011b)(Table 10).

Central Valley spring-run Chinook salmon ESU includes Chinook salmon entering the Sacramento River from March to July and spawning from late August through early October, with a peak in September. Spring-run fish in the Sacramento River exhibit an ocean-type life history, emigrating as fry, sub-yearlings, and yearlings. Central Valley spring-run Chinook salmon require cool freshwater while they mature over the summer.

Table 10. Central Valley spring-run Chinook salmon populations and selected measures of population viability.

Population	Historical Abundance[a]	Mean Number of Spawners (Range)[b]	Percent Hatchery Contribution[c]	Mean Annual Population Growth Rate (λ)[d]
Butte Creek spring-run		4,513 (67-4,513)		1.30 (1.09-1.60)
Deer Creek spring-run		1,076 (243-1,076)		1.17 (1.04-1.35)
Mill Creek spring-run		491 (203-491)		1.19 (1.00-1.47)

[a] Historical abundance for the total ESU, based on gillnet fishery catches, is estimated at about 700,000 (Fisher 1994). Individual river estimates of historical abundance not provided.

[b] Recent geometric mean number of spawners as reported by Good *et al.* 2005. Note the current geometric mean for Butte, Deer and Mill creeks are also the maximum means.

[c] Between 1967 and 1999 the Feather River Hatchery released between less than 1 million to as much as 5.5 million spring-run Chinook salmon in any given year. Returns ranged from less than 1,000 spawners to about 7,000 in the late 1980s (see Good *et al.* 2005). No other hatchery data reported.

[d] The λ calculation, provided by Good *et al.* 2005, is an estimate of the population growth rate. The 90% confidence intervals are noted in parentheses.

Status and trends. NMFS originally listed Central Valley spring-run Chinook salmon as threatened on September 16, 1999 (64 FR 50393), a classification this species retained on June 28, 2005 (70 FR 37160). This species was listed because dams isolate them from most of their historic spawning habitat and the habitat remaining to them is degraded. Historically, spring-run Chinook salmon were predominant throughout the Central Valley occupying the upper and middle reaches (1,000 to 6,000 feet) of the San Joaquin, American, Yuba, Feather, Sacramento, McCloud and Pit Rivers, with smaller populations in most tributaries with sufficient habitat for over-summering adults (Stone 1874{Rutter, 1904 #1367, Clark 1929).

The Central Valley drainage as a whole is estimated to have supported spring-run Chinook salmon runs as large as 700,000 fish between the late 1880s and the 1940s (Fisher 1994) , although these estimates may reflect an already declining population, in part from the commercial gillnet fishery that occurred in this ESU (Good *et al.* 2005b). Before construction of Friant Dam, nearly 50,000 adults were counted in the San Joaquin River alone {Fry, 1961 #1049}. Following the completion of Friant Dam, the native population from the San Joaquin River and its tributaries (i.e., the Stanislaus and Mokelumne Rivers) was extirpated. Spring-run Chinook salmon no longer exist in the American River due to the operation of Folsom Dam. Naturally spawning populations of Central Valley spring-run Chinook salmon currently are restricted to accessible reaches of the upper Sacramento River, Antelope Creek, Battle Creek, Beegum Creek, Big Chico Creek, Butte Creek, Clear Creek, Deer Creek, Feather River, Mill Creek, and Yuba River (CDFG 1998). Since 1969, the Central Valley spring-run Chinook salmon ESU (excluding Feather River fish) has displayed broad fluctuations in abundance ranging from 25,890 in 1982 to 1,403 in 1993 (CDFG unpublished data in Good *et al.* 2005b).

The average abundance for the ESU was 12,499 for the period of 1969 to 1979, 12,981 for the period of 1980 to 1990, and 6,542 for the period of 1991 to 2001. In 2003 and 2004, total run size for the ESU was 8,775 and 9,872 adults respectively, well above the 1991 to 2001 average. Evaluating the ESU as a whole, however, masks significant changes that are occurring among populations that comprise the ESU (metapopulation). For example, the mainstem Sacramento River population has undergone a significant decline while the abundance of many tributary populations increased. Average abundance of Sacramento River mainstem spring-run Chinook salmon recently declined from a high of 12,107 for the period 1980 to 1990, to a low of 609 for the period 1991 to 2001, while the average abundance of Sacramento River tributary populations increased from a low of 1,227 to a high of 5,925 over the same periods.

Abundance time series data for Mill, Deer, Butte, and Big Chico creeks spring-run Chinook salmon confirm that population increases seen in the 1990s have continued through 2001 (Good *et al.* 2005b). Habitat improvements, including the removal of several small dams and increases in summer flows in the watersheds, reduced ocean fisheries, and a favorable terrestrial and marine climate, have likely contributed to this. Up until 2001, all three spring-run Chinook salmon populations in the Central Valley have long-and short-term positive population growth. However, since 2001, escapement has declined for all three populations, possibly resulting from 2007-2009 drough conditions in California's central valley and/or unusual ocean temperatures in 2005 and 2006 (NMFS 2011b). An execption to this is the continued abundance increase at Battle and Clear Creeks (NMFS 2011b). Although the populations are small, Central Valley spring-run Chinook salmon have some of the highest population growth rates in the Central Valley.

Critical habitat. NMFS designated critical habitat for Central Valley spring-run Chinook salmon on September 2, 2005 (70 FR 52488). Specific geographic areas designated include the following CALWATER hydrological units: Tehama, Whitmore, Redding, Eastern Tehama, Sacramento Delta, Valley-Putah-Cache, Marysville, Yuba, Valley-American, Colusa Basin, Butte Creek, and Shasta Bally hydrological units. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. The critical habitat designation for this ESU identifies primary constituent elements that include sites necessary to support one or more Chinook salmon life stages. Specific sites 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. The critical habitat designation (70 FR 52488) contains additional details on the sub-areas that are included as part of this designation, and the areas that were excluded from designation.

In total, Central Valley spring-run Chinook salmon occupy 37 watersheds (freshwater and estuarine). The total area of habitat designated as critical includes about 1,100 miles of stream habitat and about 250 square miles of estuarine habitat in the San Francisco-San Pablo-Suisun Bay complex. 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 bankfull elevation. In estuarine areas the lateral extent is defined by the extreme high water because extreme high tide areas encompass those areas typically inundated by water and regularly occupied by juvenile salmon during the spring and summer, when they are migrating in the nearshore zone and relying on cover and refuge qualities provided by these habitats, and while they are foraging. Of the 37 watersheds reviewed in NMFS' assessment of critical habitat for Central Valley spring-run Chinook salmon, seven watersheds received a low rating of conservation value, three received a medium rating, and 27 received a high rating of conservation value for the species.

Upper Columbia River Chinook salmon

Distribution. The Upper Columbia River spring-run Chinook salmon ESU includes all naturally spawned populations in all river reaches accessible to Chinook salmon in Columbia River tributaries upstream of Rock Island Dam and downstream of Chief Joseph Dam in Washington, excluding the Okanogan River (Figure 7). Six artificial propagation programs are part of this 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. Spring-runs currently spawn in only three river basins above Rock Island Dam: the Wenatchee, Entiat, and Methow Rivers. Table 11 identifies ESU populations, their abundances, and estimates of the proportion of hatchery fish that contribute to run sizes.

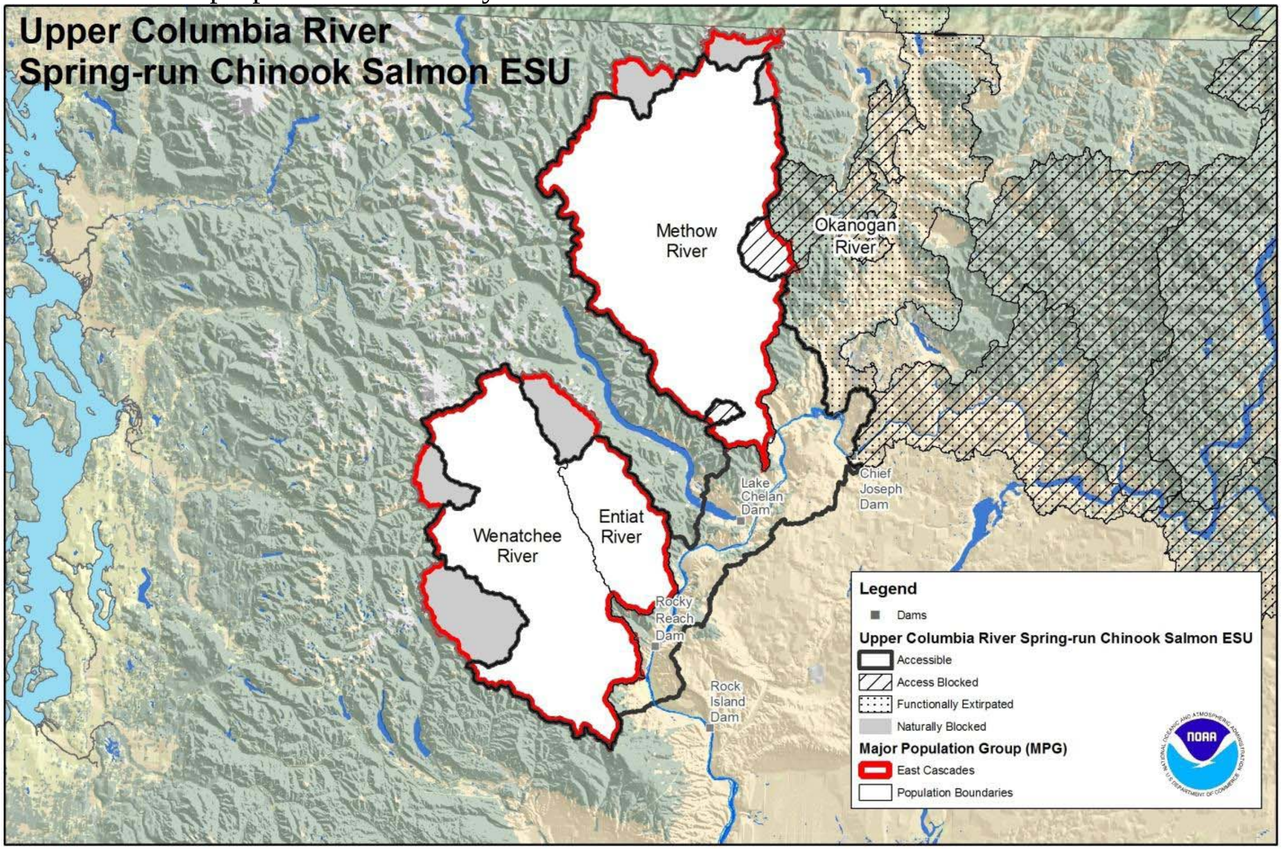

Figure 7. Population boundaries, dams, accessible areas, and extirpated reaches of upper Columbia River Chinook distribution, incorporating critical habitat boundaries (dark black line). Taken from NMFS (NMFS 2011i).

Upper Columbia River spring-run Chinook salmon begin returning to the Columbia in early spring and enter upper Columbia tributaries from April through July, with a peak in mid-May. Run timing is later when river temperature is cooler and flow is high; the corollary is also true (Keefer *et al.* 2008). After migration, individuals hold in freshwater tributaries until spawning in late summer, peaking in mid- to late August. Juvenile spring-runs remain in freshwater for a full year before emigrating to saltwater in the spring of their second year.

Table 11. Upper Columbia River Chinook salmon populations and selected measures of population viability.

Population	Mean number of spawners (range)[a]	Percent hatchery contribution[b]	Current short-term trend (previous)[c]
Methow River	680 (79-9,904)	59	+2.0 (-15.3)
Methow mainstem	161 redds (17-2,864)	59	+6.5
Twisp River	58 redds (10-369)	54	-9.8 (-27.4)
Chewuch River	58 redds (6-1,105)	41	-2.9 (-28.1)
Lost/Early Winter creeks	12 (3-164)	54	-14.1 (-23.2[d])
Entiat River	111 (53-444)	42	-1.2 (-19.4)
Wenatchee River	470 (119-4,446)	42	-1.5 (-37.4)
Chiwawa River	109 redds (34-1,046)	47	-0.7 (-29.3)
Nason Creek	54 redds (8-374)	39	-1.5 (-26.0)
Upper Wenatchee River	8 redds (0-215)	66	-8.9
White River	9 redds (1-104)	8	-6.6 (-35.9)
Little Wenatchee River	11 redds (3-74)	21	-25.8 (-25.8)

[a] 5-year geometric mean number of spawners unless otherwise noted; Includes hatchery fish. Range denoted in parentheses. Means calculated from years 1997 to 2001, except Lost/Early Winter creeks did not include 1998 as no data was available. Data reported in Good *et al.* 2005.
[b] Percent hatchery-origin from 1987-1996, and reported in Good *et al.* 2005.
[c] Current trend – percent/year – from years 1997 to 2001. Previous trend, noted in parentheses, from 1987-1996. From Good *et al.* 2005.
[d] Lost River data only.

Status and trends. NMFS listed Upper Columbia River spring-run Chinook salmon as endangered on March 24, 1999 (64 FR 14308), and reaffirmed their status as endangered on June 28, 2005 (70 FR 37160), because they had been reduced to small populations in three watersheds. Based on redd count data series, spawning escapements for the Wenatchee, Entiat, and Methow rivers have declined an average of 5.6, 4.8, and 6.3% per year, respectively, since 1958. In the most recent 5-year geometric mean (1997-2001), spawning escapement for naturally produced fish was 273 for the Wenatchee population, 65 for the Entiat population, and 282 for the Methow population, only 8-15% of the minimum abundance thresholds, although escapement increased substantially in 2000 and 2001 in all three river systems. Based on 1980-2004 returns, the average annual growth rate for this ESU is estimated at 0.93 (meaning the population is not replacing itself) (Fisher and Hinrichsen 2006). Assuming that population growth rates were to continue at 1980 to 2004 levels, Upper Columbia River spring-run Chinook salmon populations are projected to have very high probabilities of extinction within 50 years. Population viability analyses for this species (using the Dennis Model) suggest that these Chinook salmon face a significant risk of extinction: a 75 to 100% probability of extinction within 100 years (given return rates for 1980 to present). Since 2005, abundance estimates have increased throughout the ESU, although a large portion of returns are hatchery origin fish (NMFS 2011i).

Hatchery influence and genetic diversity are significant issues for the continued survival of Upper Columbia River Chinook salmon. This is a result of reduced genetic diversity from homogenization of populations that occurred under the Grand Coulee Fish Maintenance Project from 1939-1943. Stray hatchery fish and a high proportion of hatchery fish during spawning have contributed to the high genetic diversity risk.

Critical habitat. NMFS designated critical habitat for Upper Columbia River spring-run Chinook salmon on September 2, 2005 (70 FR 52630). The designation includes all Columbia

River estuaries and river reaches upstream to Chief Joseph Dam and several tributary subbasins. 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. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. The critical habitat designation for this ESU identifies primary constituent elements that include sites necessary to support one or more life stages. Specific sites 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. The Upper Columbia River spring-run Chinook salmon ESU has 31 watersheds within its range. Five watersheds received a medium rating and 26 received a high rating of conservation value to the ESU. The Columbia River rearing/migration corridor downstream of the spawning range was rated as a high conservation value. Factors contributing to the downward trends in this ESU include mainstem Columbia River hydropower system mortality, tributary riparian degradation and loss of in-river wood, altered tributary floodplain and channel morphology, reduced tributary stream flow and impaired passage, and harvest impacts.

Migratory habitat quality in this area has been severely affected by the development and operation of the FCRPS dams and reservoirs in the mainstem Columbia River, Bureau of Reclamation tributary projects, and privately owned dams in the Snake and Upper Columbia river basins. For example, construction of Hells Canyon Dam eliminated access to several likely production areas in Oregon and Idaho, including the Burnt, Powder, Weiser, Payette, Malheur, Owyhee, and Boise river basins (Good *et al.* 2005b), and Grand Coulee and Chief Joseph dams completely block anadromous fish passage on the upper mainstem Columbia River. Hydroelectric development modified natural flow regimes, resulting in higher water temperatures, changes in fish community structure leading to increased rates of piscivorous and avian predation on juvenile salmon and steelhead, and delayed migration for both adult and juveniles. Physical features of dams such as turbines also kill migrating fish. In-river survival is inversely related to the number of hydropower projects encountered by emigrating juveniles.

Similarly, development and operation of extensive irrigation systems and dams for water withdrawal and storage in tributaries have drastically altered hydrological cycles. A series of large regulating dams on the middle and upper Deschutes River affect flow and block access to upstream habitat, and have extirpated one or more populations from the Cascades Eastern Slope major population (ICTRT 2003b). Similarly, operation and maintenance of large water reclamation systems such as the Umatilla Basin and Yakima Projects have significantly reduced flows and degraded water quality and physical habitat in this domain.

Many stream reaches designated as critical habitat are over-allocated under state water law, with more allocated water rights than existing streamflow conditions can support. Withdrawal of water, particularly during low-flow periods that commonly overlap with agricultural withdrawals, often increases summer stream temperatures, blocks fish migration, strands fish, and alters sediment transport (Spence *et al.* 1996). Reduced tributary stream flow has been identified as a major limiting factor for all listed salmon and steelhead species in this area except Snake River fall-run Chinook salmon and Snake River sockeye salmon (NMFS 2007c, 2011k).

Many stream reaches designated as critical habitat are listed on the state of Oregon’s Clean Water Act section 303(d) list for water temperature. Many areas that were historically suitable rearing and spawning habitat are now unsuitable due to high summer stream temperatures. Removal of riparian vegetation, alteration of natural stream morphology, and withdrawal of water for agricultural or municipal use all contribute to elevated stream temperatures. Contaminants such as insecticides and herbicides from agricultural runoff and heavy metals from mine waste are common in some areas of critical habitat.

Species-specific threats and limitations to recovery. Several habitat and direct-effect threats exist to hinder recovery of Upper Columbia River spring-run Chinook salmon (Upper Columbia Salmon Recovery Board 2007, NMFS 2011i). Mainstem Columbia River hydropower stop or disrupt upstream and downstream fish passage, ecosystem structure and function, flows, and water quality. Floodplain connectivity and function, channel structure and complexity, riparian areas and large woody debris recruitment, stream flow, and water quality have been degraded as a result of cumulative impacts of agriculture, forestry, and development. Past introductions and persistence of non-native (exotic) fish species continues to affect habitat conditions for listed species. Degraded estuarine and nearshore marine habitat as well as harvest in Columbia River fisheriesare additional factors in the species decline.

Puget Sound Chinook salmon

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 8). 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.

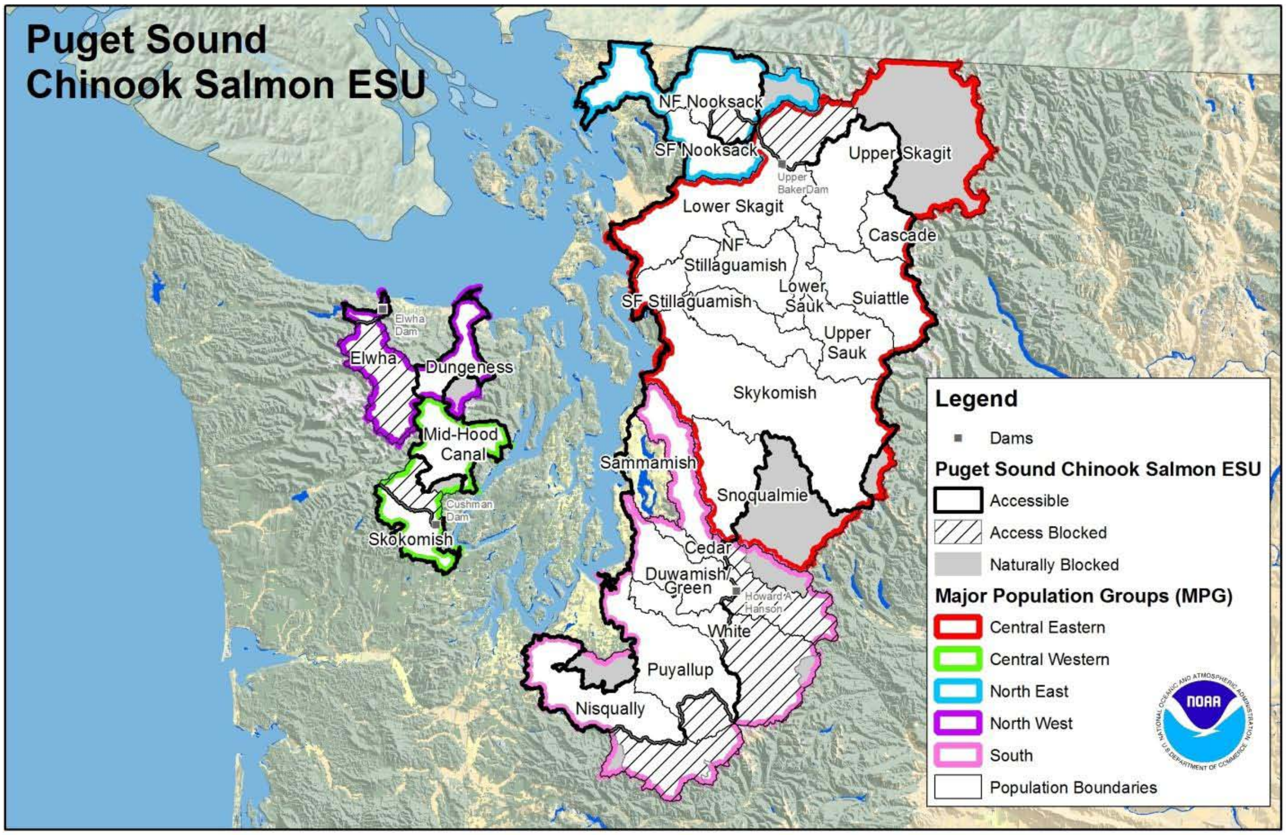

Figure 8. Population boundaries, dams, accessible areas, and extirpated reaches of Puget Sound Chinook distribution, incorporating critical habitat boundaries (dark black line). Taken from NMFS (NMFS 2011f).

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 12 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 12. 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)
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)

Population	Historical abundance[a]	Mean number of spawners (natural-origin)[b]	Percent hatchery contribution (range)[c]	λ (+/- SE)[d]
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).

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 2011f). 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).

Species-specific threats and limitations to recovery. Residential and commercial development has reduced the amount of functioning nearshore and estuarine habitat available for salmon rearing and migration (SSPS 2007, NMFS 2011f). The loss of mudflats, eelgrass meadows, and macroalgae further limits salmon foraging and rearing opportunities in nearshore and estuarine areas. Floodplain connectivity and function, channel structure and complexity, riparian areas and large wood supply, stream substrate, and water quality have also been degraded for adult spawning, embryo incubation, and rearing as a result of cumulative impacts of agriculture, forestry, and development. Total fishery exploitation rates have decreased 14 to 63% from rates in the 1980s, but weak natural-origin Chinook salmon populations in Puget Sound still require enhanced protective measures to reduce the risk of overharvest in Chinook salmon-directed fisheries.

Snake River fall-run Chinook salmon

Life history types. Although Snake River fall-run Chinook were originally listed under the assumption that all individuals migrated to the ocean at age 0 (wintering in the ocean), studies through the 1990s demonstrate that two distinct life histories actually exist (Connor *et al.* 2005b). Some individuals overwinter in the Lower Granite Reservoir and enter the ocean at age 1 and roughly twice the length of individuals entering the Pacific Ocean at age 0 (Connor *et al.* 2005b). Fish of both life histories return at roughly the same time, with 41% of wild and 61% of hatchery fish being the reservoir type (Connor *et al.* 2005b).

Distribution. The Snake River fall-run Chinook salmon ESU includes all naturally spawned populations of fall-run Chinook salmon in the mainstem Snake River below Hells Canyon Dam, and in the Tucannon River, Grande Ronde River, Imnaha River, Salmon River, and Clearwater River subbasins (Figure 9). Four artificial propagation programs are part of this 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.

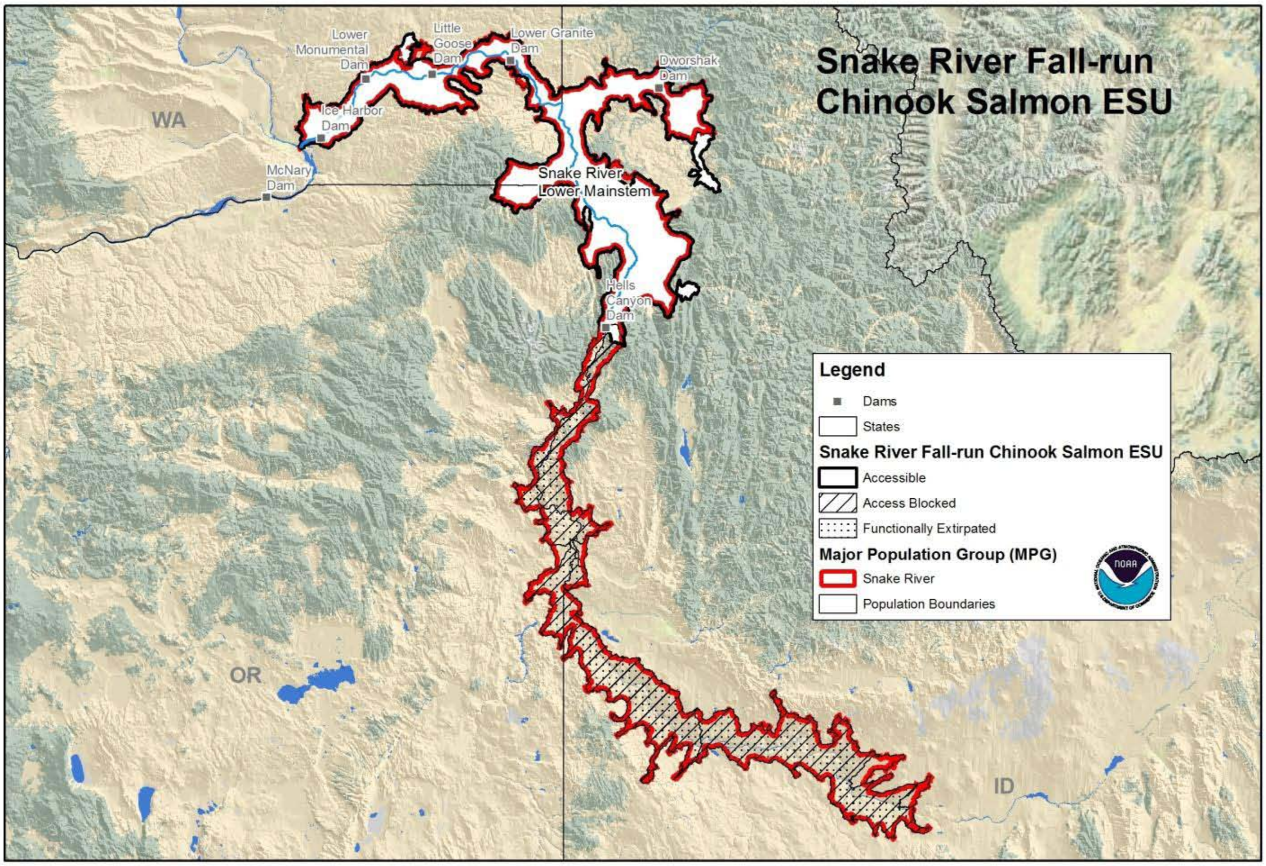

Figure 9. Population boundaries, accessible areas, extirpated reaches, and dams of Snake River fall run Chinook distribution, incorporating critical habitat boundaries (dark black line). Taken from NMFS (2011l).

Historically, the primary fall-run Chinook salmon spawning areas occurred on the upper mainstem Snake River (Connor *et al.* 2005a). A series of Snake River dams blocked access to the upper reaches, which significantly reduced spawning and rearing habitat. Consequently, salmon now reside in waters that are generally cooler than pre-dam habitats. Currently, natural spawning occurs at the upper end of Lower Granite Reservoir to Hells Canyon Dam, the lower reaches of the Imnaha, Grande Ronde, Clearwater, and Tucannon rivers, and small mainstem sections in the tailraces of the lower Snake River hydroelectric dams.

Adult Snake River fall-run Chinook salmon enter the Columbia River in July and August, and spawning occurs from October through November. Juveniles emerge from gravel in March and April of the following year, moving downstream from natal spawning and early rearing areas from June through early fall. Prior to dam construction, fall Chinook salmon were primarily ocean-type (migrated downstream and reared in the mainstem Snake River during their first year). However, today both an ocean-type and reservoir-type occur (Connor *et al.* 2005a). The reservoir-type juveniles overwinter in pools created by dams before migrating to sea; this response is likely due to early development in cooler temperatures, which prevents rapid growth. Phenotypic characteristics have shifted in apparent response to environmental changes from hydroelectric dams (Connor *et al.* 2005a). Migration downstream appears to be influenced by flow velocity within both river and reservoir systems (Tiffan *et al.* 2009).

Status and trends. The NMFS originally listed Snake River fall-run Chinook salmon as endangered in 1992 (57 FR 14653) but reclassified their status as threatened on June 28, 2005 (70 FR 37160). Estimated annual returns for the period 1938-1949 were 72,000 fish, and by the 1950s, numbers had declined to an annual average of 29,000 fish (Bjornn and Horner 1980). Numbers of Snake River fall-run Chinook salmon continued to decline during the 1960s and 1970s as approximately 80% of their historic habitat was eliminated or severely degraded by the construction of the Hells Canyon complex (1958-1967) and the lower Snake River dams (1961-1975). Counts of natural-origin adults at Lower Granite Dam were 1,000 fish in 1975, and ranged from 78 to 905 fish (with an average of 489 fish) over the ensuing 25-year period (Good *et al.* 2005b). Numbers of natural-origin individuals have increased over the last few years, with estimates at Lower Granite Dam of 2,652 fish in 2001, 2,095 fish in 2002, and 3,895 fish in 2003. Presently, hatcheries contribute rouighly 78% of the natural spawning population (NMFS 2011l).

Snake River fall-run Chinook salmon have exhibited an upward trend in returns over Lower Granite Dam since the mid 1990s. Returns classified as natural-origin spawners exceeded 2,600 fish in 2001, compared to a 1997-2001 geometric mean natural-origin count of 871 (35% of the proposed delisting abundance criteria of 2,500 natural spawners averaged over 8 years). Both the long- and short-term trends in natural returns are positive. The latest 10-year geometric mean from 199-2008 is just over 2,200 adults (NMFS 2011l). In addition, the ESU's productivity has improved to 1.28 (1990-2011) from 1.07 (1983-2003)(NMFS 2011l). Harvest impacts declined after listing and have remained relatively constant in recent years. Mainstem conditions for subyearling Chinook migrants from the Snake River have generally improved since the early 1990s. The hatchery component, derived from outside the basin, has decreased as a percentage of the run at Lower Granite Dam from the 1998/99 status reviews (5-year average of 26.2%) to 2001 (8%). This reflects an increase in the Lyons Ferry component, systematic removal of marked hatchery fish at the Lower Granite trap, and modifications to the Umatilla supplementation program to increase homing of release groups. Hatcheries stocking fish produce genetic affects in the population due to three major components: natural-origin fish (which may be progeny of hatchery fish), returns of Snake River fish from the Lyons Ferry Hatchery program, and strays from hatchery programs outside the Snake River. Current models predict a 5-25% probability of extinction within 100 years (NMFS 2011l).

Critical habitat. The NMFS designated critical habitat for Snake River fall-run Chinook salmon on December 28, 1993 (58 FR 68543). This critical habitat encompasses the waters, waterway bottoms, and adjacent riparian zones of specified lakes and river reaches in the Columbia River that are or were accessible to listed individuals (except reaches above impassable natural falls, and Dworshak and Hells Canyon Dams). These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. Adjacent riparian zones are defined as those areas within a horizontal distance of 300 feet from the normal line of high water of a stream channel or from the shoreline of a standing body of water. Designated critical habitat includes the Columbia River from a straight line connecting the west end of the Clatsop jetty (Oregon side) and the west end of the Peacock jetty (Washington side) and including all river reaches from the estuary upstream to the confluence of the Snake River, and all Snake River reaches upstream to Hells Canyon Dam. Critical habitat also includes several river reaches presently or historically accessible. Limiting factors include: mainstem lower Snake and Columbia hydrosystem mortality, degraded water quality, reduced spawning and rearing habitat due to mainstem lower Snake River hydropower system, harvest impacts, impaired stream flows, barriers to fish passage in tributaries, excessive sediment, and altered floodplain and channel morphology (NMFS 2005b).

Species-specific threats and limitations to recovery. Floodplain connectivity and function, and channel structure and complexity have been degraded as a result of cumulative impacts of agriculture, forestry, and development have hindered Snake River fall-run Chinook salmon recovery, as has degraded freshwater, estuarine, and marine habitat, harvest and hjatchery effects, as well as mainstem Columbia River and Snake River hydropower facilities (NMFS 2011l).

Snake River spring/summer-run Chinook salmon

Distribution. The Snake River spring/summer run Chinook salmon ESU includes all naturally spawned populations of spring/summer-run Chinook salmon in the mainstem Snake River and the Tucannon River, Grande Ronde River, Imnaha River, and Salmon River subbasins (Figure 10). Fifteen 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. The Interior Columbia Basin Technical Recovery Team identified 32 populations in five major population groups (Upper Salmon River, South Fork Salmon River, Middle Fork Salmon River, Grande Ronde/Imnaha, Lower Snake Mainstem Tributaries) for this species. Historic populations above Hells Canyon Dam are considered extinct (ICBTRT 2003) Table 13 identifies extant populations within the Snake River spring/summer Chinook salmon ESU, their abundances, and the relative contribution of hatchery fish.

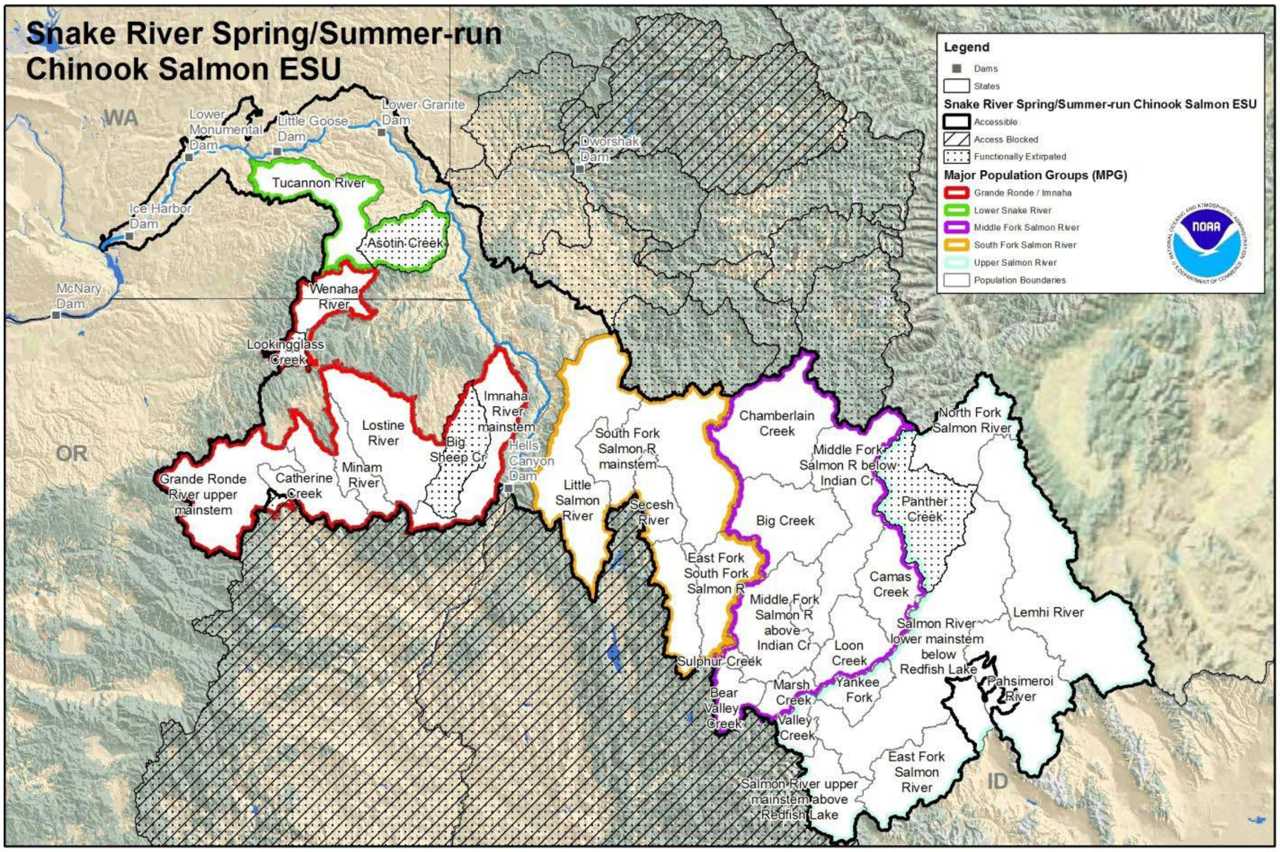

Figure 10. Population boundaries, accessible areas, extirpated reaches, and dams of Snake River spring/summer run Chinook distribution, incorporating critical habitat boundaries (dark black line). Taken from NMFS (2011l).

Snake River spring/summer-run Chinook salmon have a stream-type life history. Spawning occurs in late summer and early fall and eggs incubate over winter and hatch in late winter and early spring of the following year. Juveniles mature in the river for one year before migrating to the ocean in the spring of their second year. Larger outmigrants have a higher survival rate during outmigration (Zabel and Williams 2002, Zabel and Achord 2004). Depending on tributary and the specific habitat conditions, juveniles may migrate widely from natal reaches into alternative summer-rearing or overwintering areas. Spawners return to spawn primarily as 4- and 5-year-olds after 2 to 3 years in the ocean. A small fraction return as 3-year-old “jacks” (although sexually mature upon return, these fish are smaller in body and 1-2 years younger than most males on the spawning ground).

Table 13. Snake River spring/summer Chinook salmon populations and selected measures of population viability (rpm = redds per mile).

Current populations’	Mean number of spawners (range)[a]	Percent hatchery contribution[b]	Short-term trend (previous)[c]
Tucannon River	303 (128-1,012)	76	-4.1 (-11.0)
Wenaha River	225 (67-586)	64	-9.4 (-23.6)
Wallowa River	0.57 redds (0-29)	5	11.5
Lostine River	34 redds (9-131)	5	12.7
Minam River	180 (96-573)	5	3.3 (-14.5)
Catherine Creek	50 (13-262)	56	-25.1 (-22.5)
Upper Grande Ronde River	46 (3-336)	58	-9.4

Current populations’	**Mean number of spawners (range)[a]**	**Percent hatchery contribution[b]**	**Short-term trend (previous)[c]**
South Fork Salmon River	496 redds (277-679)	9	1.1 (-13.6)
Secesh River	144 redds (38-444)	4	9.8
Johnson Creek	131 redds (49-444)		-1.5
Big Creek spring run	53 (21-296)		5.4 (-34.2)
Big Creek summer run	5 redds (2-58)		1.7 (-27.9)
Loon Creek	27 redds (6-255)		12.2
Marsh Creek	53 (0-164)		-4.0
Bear Valley/Elk Creek	266 (72-712)		6.2
North Fork Salmon River	5.6 redds (2-19)		
Lemhi River	72 redds (35-216)		12.8 (-27.4)
Pahsimeroi River	161 (72-1,097)		12.8
East Fork Salmon spring run	0.27 rpm (0.2-1.41)		-5.7
East Fork Salmon summer run	1.22 rpm 0.35-5.32)		0.9 (-32.9)
Yankee Fork spring run	0 rpm		-6.3
Yankee Fork summer run	2.9 redds (1-18)		4.1
Valley Creek spring run	7.4 redds (2-28)		14.9 (-25.9)
Valley Creek summer run	2.14 rpm(0.71-9.29)		5.8 (-29.3
Upper Salmon spring run	69 redds (25-357)		5.3
Upper Salmon summer run	0.24 rpm (0.07-0.58)		-3.3
Alturas Lake Creek	2.7 redds (0-18)		10.2
Imnaha River	564 redds (194-3,041)	62	12.8(-24.1)
Big Sheep Creek	0.25 redds (0-1)	97	0.8
Lick Creek	1.4 redds (0-29)	59	11.7

[a] All data reported in Good *et al.* 2005. Except where noted values represent the recent geometric mean number of spawners. RPM =redds per mile.

[bc] Reported in Good *et al.* 2005.

[c] For details on data series used in calculating the population’s short term trend see Good *et al.* 2005.

Status and trends. The NMFS originally listed Snake River spring/summer-run Chinook salmon as threatened on April 22, 1992 (57 FR 14653), and reaffirmed their status as threatened on June 28, 2005 (70 FR 37160). Although direct estimates of historical annual Snake River spring/summer Chinook salmon returns are not available, returns may have declined by as much as 97% between the late 1800s and 2000. According to Matthews and Waples (1991), total annual Snake River spring/summer Chinook salmon production may have exceeded 1.5 million adult fish in the late 1800s. Total (natural plus hatchery origin) returns fell to roughly 100,000 spawners by the late 1960s and were below 10,000 by 1980 (Fulton 1968). Between 1981 and 2000, total returns fluctuated between extremes of 1,800 and 44,000 fish. The 2001 and 2002 total returns increased to over 185,000 and 97,184 adults, respectively. The 1997-2001 geometric mean total return for the summer run component at Lower Granite Dam was slightly more than 6,000 fish, compared to the geometric mean of 3,076 fish for the years 1987-1996. The 2002-2006 geometric mean of the combined Chinook salmon runs at Lower Granite Dam was over 18,000 fish. However, it is important to note that over 80% of the 2001 return and over 6% of the 2002 return originated in hatcheries (Good *et al.* 2005b). Good *et al.* (2005b) reported that risks to individual populations within the ESU may be greater than the extinction risk for the entire ESU due to low levels of annual abundance and the extensive production areas within the Snake River basin. Recent spawner returns have been higher, but low natural spawner abundance and the ability of the populations to cope with low ocean survival remain concerns (NMFS 2011l).

Critical habitat. The NMFS designated critical habitat for Snake River spring/summer-run Chinook salmon on October 25, 1999 (64 FR 57399). This critical habitat encompasses the waters, waterway bottoms, and adjacent riparian zones of specified lakes and river reaches in the Columbia River that are or were accessible to listed Snake River salmon (except reaches above impassable natural falls, and Dworshak and Hells Canyon Dams). Adjacent riparian zones are defined as those areas within a horizontal distance of 300 feet from the normal line of high water of a stream channel or from the shoreline of a standing body of water. Designated critical habitat includes the Columbia River from a straight line connecting the west end of the Clatsop jetty (Oregon side) and the west end of the Peacock jetty (Washington side) and including all river reaches from the estuary upstream to the confluence of the Snake River, and all Snake River reaches upstream to Hells Canyon Dam; the Palouse River from its confluence with the Snake River upstream to Palouse Falls, the Clearwater River from its confluence with the Snake River upstream to its confluence with Lolo Creek; the North Fork Clearwater River from its confluence with the Clearwater river upstream to Dworshak Dam. Critical habitat also includes several river reaches presently or historically accessible. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. Limiting factors identified for this species include hydrosystem mortality, reduced stream flow, altered channel morphology and floodplain, excessive fine sediment, and degraded water quality (NMFS 2006i).

Species-specific threats and limitations to recovery. As with other listed salmonids, floodplain connectivity and function, channel structure and complexity, riparian areas and large wood supply, stream substrate, elevated water temperature, stream flow, and water quality have been degraded as a result of cumulative impacts of agriculture, forestry, and development to the degree that species recovery is impaired (NMFS 2006i). Additional threats to recovery exist from mainstem Columbia River and Snake River hydropower, predation, as well as harvest-related effects.

Upper Willamette River Chinook salmon

Distribution. The Upper Willamette River Chinook salmon ESU includes all naturally spawned populations of spring-run Chinook salmon in the Clackamas River and in the Willamette River, and its tributaries, above Willamette Falls, Oregon (7 populations in all)(NMFS 2011j)(Figure 11). Seven 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.

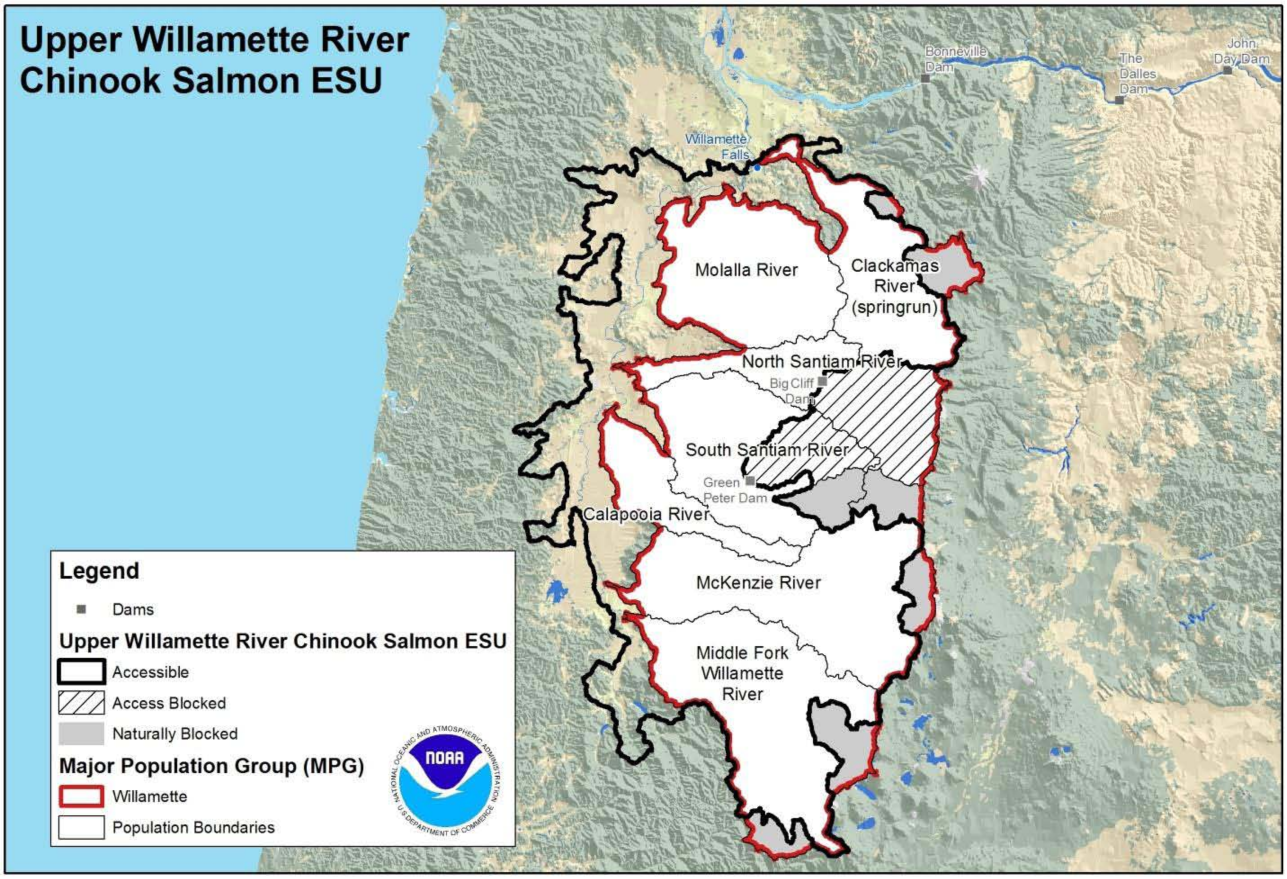

Figure 11. Population boundaries, dams, accessible areas, and extirpated reaches, of upper Willamette River Chinook distribution, incorporating critical habitat boundaries (dark black line). Taken from NMFS (2011).

Upper Willamette River Chinook salmon occupy the Willamette River and its tributaries. All spring-run Chinook salmon in the ESU, except those entering the Clackamas River, must pass Willamette Falls. In the past, this ESU included sizable numbers of spawning salmon in the Santiam River, the middle fork of the Willamette River, and the McKenzie River, as well as smaller numbers in the Molalla River, Calapooia River, and Albiqua Creek. Historically, access above Willamette Falls was restricted to spring when flows were high. In autumn, low flows prevented fish from ascending past the falls. The Upper Willamette spring-run Chinook salmon are one of the most genetically distinct Chinook salmon groups in the Columbia River Basin. Individuals enter the Columbia River and its estuary earlier than other spring Chinook salmon ESUs (Meyers *et al.* 1998). Fall-run Chinook salmon spawn in the Upper Willamette but are not considered part of the ESU because they are not native.

Status and trends. The NMFS originally listed Upper Willamette River Chinook salmon as threatened on March 24, 1999 (64 FR 14308), and reaffirmed their status on June 28, 2005 (70 FR 37160). The total abundance of adults (hatchery-origin plus natural-origin fish) passing Willamette Falls has remained relatively steady over the past 50 years (ranging from approximately 20,000 to 70,000 fish), but it is an order of magnitude below the peak abundance levels observed in the 1920s (approximately 300,000 adults). Although the Clackmas population consisted of roughly 12,000 spawners in 2004, that number dropped to 2,000 in 2009 and 2010, with a geometric mean of 850 individuals from 2006-2011 (NMFS 2011j). McKenzie River returns had been increasing in abundance until peaking in 2004, but have since dropped to historical levels (NMFS 2011j). Until recent years, interpretation of abundance levels has been confounded by a high but uncertain fraction of hatchery-produced fish. Although the number of adults crossing Willamette Falls is in the same range (about 20,000-70,000 adults) it has been for the last 50 years, a large fraction of these are hatchery produced. Estimates of the percentage of hatchery fish range according to tributary, but exceed 70% (Good *et al.* 2005b). The Calapooia River is estimated to contain 100% hatchery fish. Insufficient information on hatchery production in the past prevents a meaningful analysis of the population trend; therefore no formal trend analysis is available.

Most natural spring Chinook salmon populations of the Upper Willamette River are likely extirpated or nearly so, with only one remaining naturally reproducing population identified in the McKenzie River. Unfortunately, recent short-term declines in abundance suggest that this population may not be self-sustaining (Meyers *et al.* 1998, Good *et al.* 2005b). Abundance in this population has been relatively low (low thousands) with a substantial number of these fish being of hatchery origin. The population increased substantially from 2000 to 2003, probably due to increased survival in the ocean. Future oceanic survival rates are unpredictable, and the likelihood of long-term sustainability for this population has not been determined. Of concern is that a majority of the spawning habitat and approximately 30-40% of total historical habitat are no longer accessible because of dams (Good *et al.* 2005b). Individuals from the ESU migrate far north and are caught incidentally in ocean fisheries, particularly off southeast Alaska and northern Canada, and in the mainstem Columbia and Willamette rivers during spring.

Critical habitat. The NMFS designated critical habitat for Upper Willamette River Chinook salmon on September 2, 2005 (70 FR 52630). Critical habitat includes defined areas within subbasins of the middle fork Willamette River, upper Willamette River, McKenzie River, Santiam River, Crabtree Creek, Molalla River, and Clackamas River. 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 critical habitat designation for this ESU identifies primary constituent elements that include sites necessary to support one or more Chinook salmon life stages. Specific sites include freshwater spawning and rearing sites, freshwater migration corridors. The physical or biological features that characterize these sites include water quality and quantity, natural cover, forage, adequate passage conditions, and floodplain connectivity. Of 65 subbasins reviewed in NMFS' assessment of ESU critical habitat, 19 subbasins were rated as having a medium conservation value, 19 were rated as low, and the 27 remaining subbasins were rated as having a high conservation value. Federal lands were generally rated as having high conservation value to the species' spawning and rearing. Factors contributing to the downward trends in this ESU include reduced access to spawning/rearing habitat in tributaries, hatchery impacts, altered water quality and temperature in tributaries, altered stream flow in tributaries, and lost or degraded floodplain connectivity and lowland stream habitat.

Land management activities have severely degraded stream habitat conditions in the Willamette River mainstem above Willamette Falls and associated subbasins. In the Willamette River mainstem and lower sub-basin mainstem reaches, high density urban development and widespread agricultural effects have reduced aquatic and riparian habitat quality and complexity, and altered sediment and water quality and quantity, and watershed processes. The Willamette River, once a highly braided river system, has been dramatically simplified through channelization, dredging, and other activities that have reduced rearing habitat by as much as 75%. In addition, the construction of 37 dams in the basin blocked access to more than 435 miles of stream and river spawning habitat. The dams alter the temperature regime of the Willamette River and its tributaries, affecting the timing and development of naturally-spawned eggs and fry. Agriculture, urbanization, and gravel mining on the valley floor logging in the Cascade and Coast Ranges contribute to increased erosion and sediment loads throughout the basin.

The mainstem Willamette River has been channelized and stripped of large wood. Development began to encroach on the riparian forest beginning in the 1870s (Sedell and Froggatt 1984). Gregory *et al.* (2002d)(calculated that the total mainstem Willamette River channel area decreased from 41,000 to 23,000 acres between 1895 and 1995. They noted that the lower reach, from the mouth of the river to Newberg (RM 50), is confined within a basaltic trench, and that due to this geomorphic constraint, less channel area has been lost than in upstream areas. The middle reach from Newberg to Albany (RM 50 to 120) incurred losses of 12% primary channel area, 16% side channels, 33% alcoves, and 9% islands. Even greater changes occurred in the upper reach, from Albany to Eugene (RM 187). There, approximately 40% of both channel length and channel area were lost, along with 21% of the primary channel, 41% of side channels, 74% of alcoves, and 80% of island areas.

The banks of the Willamette River have more than 96 miles of revetments; approximately half were constructed by the ACOE. Generally, the revetments were placed in the vicinity of roads or on the outside bank of river bends, so that while only 26% of the total length is revetted, 65% of the meander bends are revetted (Gregory *et al.* 2002c). The majority of dynamic sections have been armored, reducing adjustments in channel bed and sediment storage by the river, and thereby diminishing both the complexity and productivity of aquatic habitats (Gregory *et al.* 2002b).

Riparian forests have diminished considerably in the lower reaches of the Willamette River (Gregory *et al.* 2002a). Sedell and Froggatt (1984) noted that agriculture and cutting of streamside trees were major agents of change for riparian vegetation, along with snagging of large wood in the channel. The reduced shoreline, fewer and smaller snags, and reduced riparian forest comprise large functional losses to the river, reducing structural features, organic inputs from litter fall, entrained allochthonous materials, and flood flow filtering capacity. Extensive changes began before the major dams were built, with navigational and agricultural demands dominating the early use of the river. The once expansive forests of the Willamette River floodplain provided valuable nutrients and organic matter during flood pulses, food sources for macroinvertebrates, and slow-water refugia for fish during flood events. These forests also cooled river temperatures as the river flowed through its many channels.

Gregory *et al.* (2002a) described the changes in riparian vegetation in river reaches from the mouth to Newberg, from Newberg to Albany, and from Albany to Eugene. They noted that the riparian forests were formerly a mosaic of brush, marsh, and ash tree openings maintained by annual flood inundation. Below the City of Newberg, the most noticeable change was that conifers were almost eliminated. Above Newberg, the formerly hardwood-dominated riparian forests along with mixed forest made up less than half of the riparian vegetation by 1990, while agriculture dominated. This conversion has reduced river shading and the potential for recruitment of wood to the river, reducing channel complexity and the quality of rearing, migration and spawning habitats.

Hyporheic flow in the Willamette River has been examined through discharge measurements and found to be significant in some areas, particularly those with gravel deposits (Wentz *et al.* 1998, Fernald *et al.* 2001). The loss of channel complexity and meandering that fosters creations of gravel deposits decreases the potential for hyporheic flows, as does gravel mining. Hyporheic flow processes water and affects its quality on reemerging into the main channel, stabilizing variations in physical and chemical water characteristics. Hyporheic flow is important for ecological functions, some aspects of water quality (such as temperature and dissolved oxygen), and some benthic invertebrate life stages. Alcove habitat, which has been limited by channelization, combines low hydraulic stress and high food availability with the potential for hyporheic flows across the steep hydraulic gradients in the gravel separating them from the main channel (Fernald *et al.* 2001).

Species-specific threats and limitations to recovery. Upper Willamette River Chinook recovery is specifically being hindered by several factors, including reduced access to spawning and rearing habitat because of tributary dams, degraded freshwater habitat, especially floodplain connectivity and function, channel structure and complexity, and riparian areas and large wood recruitment as a result of cumulative impacts of agriculture, forestry, and development, degraded water quality and altered temperature as a result of both tributary dams and the cumulative impacts of agriculture, forestry, and urban development, hatchery-related effects, and ccean harvest rates of approximately 30 percent (NMFS 2011j, ODFW and NMFS 2011). Significantly, anthropogenic introductions of non-native species and out-of-ESU races of salmon or steelhead have increased predation on, and competition with, native Upper Willamette River Chinook salmon to the point of hindering recovery of the species.

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.

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.* 1991c, 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 (Simenstad and Salo 1982, Salo 1991). 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.

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 2011l). 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 2011l).

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.

Columbia River chum salmon

Distribution. The Columbia River chum ESU includes all naturally-spawned populations of chum salmon in the Columbia River and its tributaries in Washington and Oregon (Figure 12). Three 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.

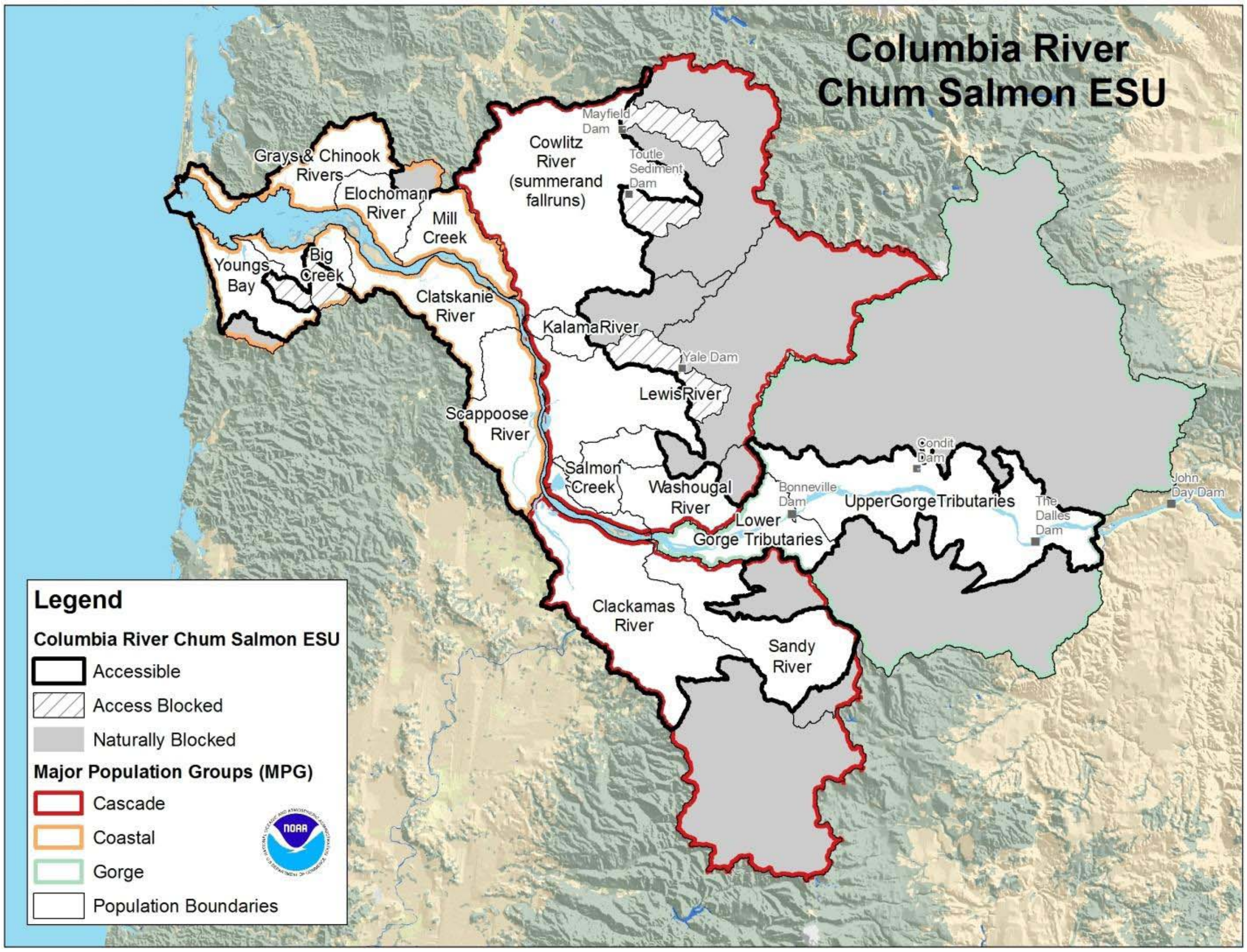

Figure 12. Population boundaries, dams, accessible areas, and extirpated reaches of Columbia River chum distribution, incorporating critical habitat boundaries (dark black line). Taken from NMFS (2011).

Most of the chum within this ESU return to northern tributaries of the Columbia River (in Washington State), primarily the Grays River, in areas immediately below Bonneville Dam, and in smaller numbers under the I-205 bridge near Vancouver. Chum populations that formerly occupied tributaries on the south bank of the Columbia (in Oregon) are considered extirpated or nearly so. Observers have documented spawning over multiple years in the mainstem Columbia River, near McCord Creek and Multnomah Falls in Oregon, although the number of spawners in these areas are generally quite low (McElhany *et al.* 2007). Chum salmon return to the Columbia River in late fall (mid-October to December).

Status and trends. The NMFS listed Columbia River chum salmon as threatened on March 25, 1999, and reaffirmed their status on June 28, 2005 (71 FR 37160). Chum salmon in the Columbia River once numbered in the hundreds of thousands of adults and were reported in almost every river in the Lower Columbia River basin, but by the 1950s most runs disappeared (Rich 1942, Marr 1943, Fulton 1970). The total number of chum salmon returning to the Columbia River in the last 50 years has averaged a few thousand per year, with returns limited to a very restricted portion of the historical range. Significant spawning occurs in only two of the 17 historical populations (Table 14), meaning that 88% of the historical populations are extirpated, or nearly so. The two remaining populations are the Grays River and the lower Columbia Gorge tributaries (Good *et al.* 2005b). Both long- and short-term trends for Grays River abundance are negative, but the current trend in abundance for the lower Columbia Gorge tributaries is slightly positive. Chum salmon appear to be extirpated from the Oregon portion of this ESU. In 2000, ODFW conducted surveys to determine the abundance and distribution of chum salmon in the Columbia River, and out of 30 sites surveyed, only one chum salmon was observed.

Table 14. Columbia River chum salmon populations and selected measures of population viability.

Current populations	Historical abundance[a]	Recent spawner abundance	Short-term median growth rate (λ)[c]
Youngs Bay			
Gray's River	7,511	331/704[b]	1.043 (0.957-1.137)
Big Creek			
Elochoman River			
Clatskanie River			
Mill, Abernathy, and Germany Creeks			
Scappoose Creek			
Cowlitz River	141,582		
Kalama River	9,953		
Lewis River	89,671		
Salmon Creek			
Clackamus River			
Sandy River			
Washougal River	15,140		
Lower gorge tributaries	>3,141	425[b]	0.984 (0.883-1.096)
Upper gorge tributaries	>8,912		

[a] Estimated total historical abundance for this ESU was about 283,421 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 using equilibrium abundance under historical conditions. All data are reported in Good *et al.* 2005.
[b] Two different time series estimate are available but based on overlapping years. The first estimate is based on 1996-2000 data, while the second is based on 1996-2000 data.
[c] The λ calculation is an estimate of what the natural growth rate would have been after accounting for hatchery-origin spawners. Two different estimates are available for the Grays River population; the Rawlings estimate (depicted in the table above) is believed to be more accurate. Other estimates, long- and short-term trends, suggest the population is declining (see Good *et al.* 2005).

Few Columbia River chum salmon have been observed in tributaries between The Dalles and Bonneville dams. Surveys of the White Salmon River in 2002 found one male and one female carcass, with no evidence of spawning (Ehlke and Keller 2003). Chum salmon were not observed in any upper Columbia Gorge tributaries during the 2003 and 2004 spawning ground surveys. Most Columbia River chum populations have been functionally extirpated or are presently at very low abundance levels. Outmigrating chum fry were observed moving past Bonneville Dam in 2010 for the first time (NMFS 2011d).

Historically, Columbia River chum salmon supported a large commercial fishery in the first half of this century, which landed more than 500,000 fish per year as recently as 1942. Commercial catches declined beginning in the mid-1950s, and in later years rarely exceeded 2,000 per year. During the 1980s and 1990s, the combined abundance of natural spawners for the lower Columbia Gorge, Washougal, and Grays River populations was below 4,000 adults. In 2002, however, the abundance of natural spawners exhibited a substantial increase at several locations (estimate of natural spawners is approximately 20,000 adults). The cause of this dramatic increase in abundance is unknown. However, long- and short-term productivity trends for populations are at or below replacement. The loss of off-channel habitat and the extirpation of approximately 17 historical populations increase this species' vulnerability to environmental variability and catastrophic events. Overall, the populations that remain have low abundance, limited distribution, and poor connectivity (Good *et al.* 2005b).

Critical habitat. The NMFS designated critical habitat for Columbia River chum salmon on September 2, 2005 (70 FR 52630). The designated includes defined areas in the following subbasins: Middle Columbia/Hood, Lower Columbia/Sandy, Lewis, Lower Columbia/Clatskanie, Lower Cowlitz, Lower Columbia subbasin and river corridor. 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 critical habitat designation for this ESU identifies primary constituent elements that include sites necessary to support one or more chum salmon life stages. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding and are rated as having high conservation value to the species. Columbia River chum salmon have primary constituent elements of freshwater spawning, freshwater rearing, freshwater 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 21 subbasins reviewed in NMFS' assessment of critical habitat for the ESU, three subbasins were rated as having a medium conservation value, no subbasins were rated as low, and 18 subbasins, were rated as having a high conservation value. The major factors limiting recovery are altered channel form and stability in tributaries, excessive sediment in tributary spawning gravels, altered stream flow in tributaries and the mainstem Columbia River, loss of some tributary habitat types, and harassment of spawners in the tributaries and mainstem.

Species-specific threats and limitations to recovery. Lower Columbia River Chinook salmon face numerous hindrances to their survival and recovery, including degraded estuarine and near-shore marine habitat resulting from cumulative impacts of land use and flow management by the Columbia River hydropower system and floodplain connectivity and function, channel structure and complexity, riparian areas, stream substrate, stream flow, and water quality have been degraded as a result of cumulative impacts of agriculture, hydropower, forestry, and development (LCFRB 2010, NMFS 2011d). Further habitat-based impacts to the species result from an altered flow regime and Columbia River plume has altered the temperature regime and estuarine food web, reduced water quality, and has reduced ocean productivity, reduced access to off-channel rearing habitat in the lower Columbia River , loss of access and loss of some habitat types as a result of passage barriers such as roads and railroads, reduced access to off-channel rearing habitat in the lower Columbia River, and reduced productivity resulting from sediment and nutrient-related changes in the estuary. Also hampering the species is reduced access to spawning and rearing habitat mainly as a result of tributary hydropower projects, current or potential predation from hatchery-origin salmonids, including coho salmon, juvenile fish strandings that result from ship wakes, and contaminants affecting fish health and reproduction.

Hood Canal summer-run chum salmon

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 13) 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 15 identifies populations within the Hood Canal summer-run chum salmon ESU, their abundances, and hatchery input.

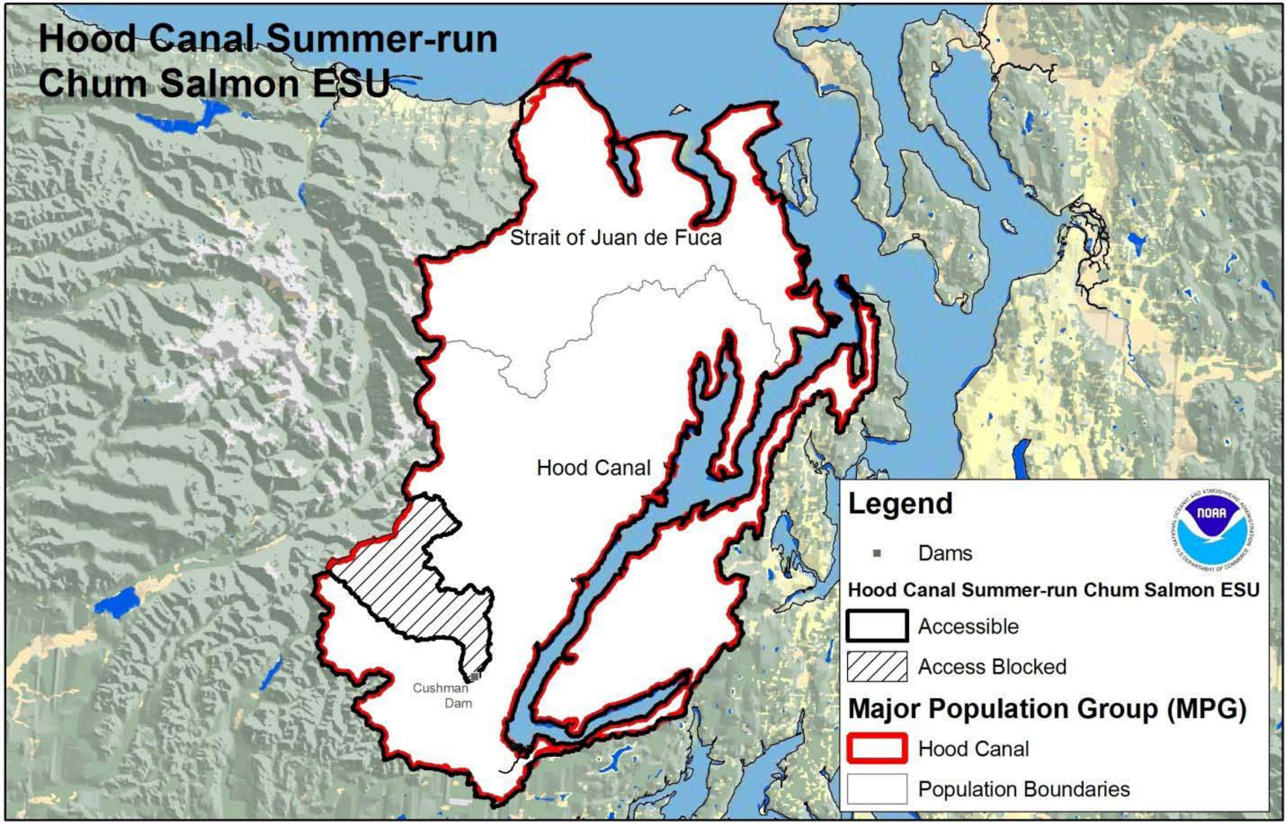

Figure 13. Population boundaries, dams, accessible areas, and extirpated reaches of Hood Canal summer-run chum distribution, incorporating critical habitat boundaries (dark black line). Taken from NMFS (NMFS 2011f).

Table 15. 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.

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 2011f).

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 hightide 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).

Species-specific threats and limitations to recovery. Nearshore and estuarine habitat throughout the range of the species has been altered by human activities (Brewer *et al.* 2007, NMFS 2011f). Nutrient loading has lowered dissolved oxygen concentrations, which can kill or stress marine organisms, including salmon. Residential and commercial development has reduced the amount of functioning habitat available for salmon rearing and migration. The loss of mudflats, eelgrass meadows, and macroalgae further limits salmon foraging and rearing opportunities in nearshore and estuarine areas. In addition, floodplain connectivity and function, channel structure and complexity, riparian areas and large wood supply, stream substrate, and stream flow have been degraded as a result of cumulative impacts of agriculture, forestry, and development.

Coho salmon

Description of the species. Coho salmon occur naturally in most major river basins around the North Pacific Ocean from central California to northern Japan (Laufle *et al.* 1986). The typical life history of Coho salmon is similar to most of the other large bodied Pacific salmonids, in so much as adult fish spawn in the fall and winter, young emerge in the spring, rear in freshwater and saltwater and return to spawn as adults. Sympatric in many river basins with Chinook, chum, sockeye, and pink salmon, partitioning occurs through the species' use of different areas of a river for reproduction and rearing, and the length of time they spend in these ecosystems. For instance, Chinook salmon spawn in fast flowing mainstem riverine reaches with large substrate; sockeye salmon spawn in rivers and lakes, and chum salmon spawn in mid- to lower reaches of rivers and have been observed spawning in areas of tidal influence. Coho salmon characteristically spawn in tributaries and slow-flowing shallow creeks in tributaries with gradients of 3% or less, which may be fed by cool groundwater sources, and are often widely dispersed within watersheds. Adult Coho salmon may remain in freshwater three or more months before spawning, with early migrants often moving farther upstream (Sandercock 1991).

Most Coho salmon enter rivers between September and February, but entry is influenced by discharge and other factors. In many river systems, Coho salmon are unable to enter the rivers until sufficiently strong flows open passages and provide sufficient depth. First fall freshets combined with high tides trigger the upstream migration of Coho salmon in many basins. Until then, if river flows are low or warm summer temperatures persist, fish may congregate in pools near the mouth of the river or natal stream until conditions change. Typically Coho salmon spawn from November to January, although there are many exceptions throughout their range. Spawning duration usually spans about three months in most basins, with individual fish actively spawning for several days to weeks. Spawning occurs in a few third-order streams, but most spawning activity occurs in fourth- and fifth-order streams. As with other Pacific salmon, Coho salmon fecundity varies with the size of the fish and latitudinally with Coho salmon in northern climes generally exhibiting higher rates of fecundity (Sandercock 1991). Most Coho salmon mature and spawn at age 3, although there are exceptions; in many basins in the northern portion of the species range Coho salmon spawn at age 2.

Habitat. Coho salmon are found in depths ranging from the surface to 250 m, but individuals in the open ocean generally stay within 30 m of the surface (Emmett *et al.* 1991c). Juveniles occur at even shallower depths (<10 m)(PFMC 2000). Juveniles are also found closer to shore; generally within 74 km (PFMC 2000, NMFS-NWR 2005). However, adults have been tracked well beyond the EEZ off Oregon (PFMC 2000). In years of weak upwelling, individuals tend to concentrate over submarine canyons and areas of more permanent upwelling, while strong upwelling years result in more dispersed stocks. Acceptable temperature regimes run from 4° to

15.2° C, but optimal range is between 8° and 12° C (Emmett *et al.* 1991c).

Rates of incubation are largely temperature dependent: colder water temperatures will slow development. Generally, in optimal temperatures eggs incubate for about 35-50 days, and fry start emerging from the gravel two to three weeks after hatching. Incubation and emergence success are also influenced by dissolved oxygen levels, sediment loading, and scouring high flows. Following emergence, fry aggregate and move to shallow areas near the stream banks. Most Coho salmon rear in freshwater for about 15-18 months. As fry grow, they disperse up- and downstream to establish and defend territories. Juvenile rearing usually occurs in tributaries with gradients of 3% or less, although they may move to streams with gradients of 4-5%. Juvenile Coho salmon are often found in small streams less than five feet wide, and may migrate considerable distances to rear in lakes and off-channel ponds. During the summer, fry prefer pools featuring adequate cover such as large woody debris, undercut banks, and overhanging vegetation. Overwintering tends to occur in larger pools, backwater areas, and off stream channels and ponds (e.g., wall-based channels that are groundwater fed).

At not quite 2 years of age, Coho salmon will migrate downstream where they undergo the physiological transition to salt water. The outmigration of smolts begins as early as February and may continue through the summer and fall, with peak outmigration often between March and June, although this varies among basins and environmental conditions (Sandercock 1991). Several weeks are spent in coastal waters prior to northward migration (PFMC 2000). This is particularly true for Coho originating from Oregonian streams, whose northward movement is generally delayed by strong southerly currents which weaken in the winter months (PFMC 2000). Once in the ocean, Coho salmon generally migrate north along the coast in a narrow coastal band that broadens in southeastern Alaska. During this migration, juvenile Coho salmon tend to occur in both coastal and offshore waters. During spring and summer, Coho salmon will forage in waters between 46º N, the Gulf of Alaska, and along Alaska's Aleutian Islands (PFMC 2000).

Feeding. Coho salmon are opportunistic feeders. While at sea, Coho salmon tend to eat fish, including herring, sand lance, sticklebacks, sardines, shrimp and surf smelt (Emmett *et al.* 1991c). While in estuaries and in freshwater Coho salmon are significant predators of Chinook, pink, and chum salmon, as well as aquatic and terrestrial insects. Smaller fish, such as fry, eat chironomids, plecopterans, and other larval insects, and typically use visual cues to find their prey. Juveniles appear to prefer to feed in upwelled oceanic waters, although they are also present in eddy systems (Pool *et al.* 2008). Oceanic juveniles commonly feed upon euphausiids, chaetognaths, and decapod megalopae in these locations (Pool *et al.* 2008, Brodeur *et al.* 2010).

Threats. Coho salmon, like other salmon, are exposed to high rates of natural predation at each life stage. Most mortality, however, occurs in freshwater. Winter mortality may be significant for Coho salmon because they overwinter in freshwater, where they can be swept downstream from freshets or eaten by raccoon, cutthroat trout, or other small animals. Once Coho reach the ocean, survival is high (Sandercock 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 2011l). 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 2011l).

Coho 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 Coho 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 fresh water, estuarine, and coastal ecosystems throughout the species range.

Central California Coast Coho salmon

Distribution. The Central California Coast coho salmon ESU extends from Punta Gorda in northern California south to and including the San Lorenzo River in central California (Weitkamp *et al.* 1995a). The ESU includes all naturally spawned populations of coho salmon from Punta Gorda in northern California south to and including the San Lorenzo River in central California, as well as populations in tributaries to San Francisco Bay, excluding the Sacramento-San Joaquin River system. Two natural populations have been genetically identified (NMFS 2011a). Four artificial propagation programs are part of the Central California Coast coho salmon ESU: the Don Clausen Fish Hatchery Captive Broodstock Program, Scott Creek/King although 12 independent and 63 dependent populations once existsed (NMFS 2011a). Fisher Flats Conservation Program, Scott Creek Captive Broodstock Program, and the Noyo River Fish Station egg-take Program (recently terminated) coho hatchery programs. 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.

Coho salmon in this ESU enter rivers to spawn very late (peaking in January), with little time spent in fresh water between river entry and spawning. This compressed adult freshwater residency appears to coincide with the single, brief peak of river flow characteristic of this region.

Status and trends. NMFS originally listed the central California coast coho salmon ESU as threatened on October 31, 1996 (61 FR 56138) and later reclassified their status to endangered June 28, 2005 (70 FR 37160). Information on the abundance and productivity trends for the naturally spawning component of the central California coast coho ESU is extremely limited. There are no long-term time series of spawner abundance for individual river systems. The best available data are from Lagunitas Creek, where red data have been collected since 1997 (NMFS 2011a). Here, red abundance has been declining since a peak in 2004 (NMFS 2011a). Historical estimated escapement for this ESU is 56,100 for 1963, and more recent estimates suggest the ESU dropped to about one-fourth that size by the late 1980s and early 1990s (Good *et al.* 2005c).

Where data are available, analyses of juvenile coho presence-absence information, juvenile density surveys, and irregular adult counts for the South Fork Noyo River indicate low abundance and long-term downward trends for the naturally spawning populations throughout the ESU (Good *et al.* 2005c, NMFS 2011a). Some dependent populations have recent estimates since 1999 or 2000, including Pudding Creek (495 spawners), Casper Creek (155 spawners), Little Creek (40 spawners), Redwood Creek (0 to 93 redds and on a declining trend), and Scott Creek (declining trend since 2003-2004 with a couple to few hundred spawners in these years; almost all were hatchery origin) (NMFS 2011i). Improved ocean conditions coupled with favorable stream flows and harvest restrictions have contributed to increased returns in 2001 in streams in the northern portion of the ESU, as indicated by an increase in the observed presence of fish in historically occupied streams. Data are particularly lacking for many river basins in the southern two-thirds of the ESU where naturally spawning populations are considered to be at the greatest risk. The extirpation or near extirpation of natural coho salmon populations in several major river basins, and across most of the southern historical range of the ESU, represents a significant risk to ESU spatial structure and diversity (Good *et al.* 2005c).

Artificial propagation of coho salmon within the Central California Coast ESU has declined since the ESU was listed in 1996 though it continues at the Noyo River and Scott Creek facilities, and two captive broodstock populations have recently been established. Genetic diversity risk associated with out-of-basin transfers appears to be minimal, but diversity risk from domestication selection and low effective population sizes in the remaining hatchery programs remains a concern. An out-of-ESU artificial propagation program for coho was operated at the Don Clausen hatchery on the Russian River through the mid 1990s, but was terminated in 1996. Termination of this program was considered by the biological review team as a positive development for naturally produced coho in this ESU.

For the naturally spawning component of the ESU, the biological review team found very high risk of extinction for the abundance, productivity, and spatial structure of the Viable Salmon Population (VSP) parameters and comparatively moderate risk with respect to the diversity VSP parameter. The lack of direct estimates of the performance of the naturally spawned populations in this ESU, and the associated uncertainty this generates, was of specific concern to the biological review team. Informed by the VSP risk assessment and the associated uncertainty, the strong majority opinion of the biological review team was that the naturally spawned component of the Central California Coast coho ESU was “in danger of extinction.” The minority opinion was that this ESU is “likely to become endangered within the foreseeable future” (70 FR 37160). Accordingly, NMFS upgraded the status of central California coast coho ESU to endangered on June 28, 2005 (70 FR 37160).

Central California Coast coho salmon populations continue to be depressed relative to historical numbers. Strong indications show that breeding groups have been lost from a significant percentage of historical stream range. Overall, available population trends indicate poor adult returns within the ESU between 2006 and 2010 (NMFS 2011i). A number of coho populations in the southern portion of the range appear to be either extinct or nearly so, including those in Gualala, Garcia, and Russian rivers, as well as smaller coastal streams in and south of San Francisco Bay (Good *et al.* 2005c, NMFS 2011a). However, juveniles were recently identified in Soquel Creek south of the San Lorenzo River, indicating a recolonization of that area in 2007-2008 (NMFS 2011a).

Critical habitat. NMFS designated critical habitat for central California coast coho salmon on May 5, 1999 (64 FR 24049). The designation encompasses accessible reaches of all rivers (including estuarine areas and riverine reaches) between Punta Gorda and the San Lorenzo River (inclusive) in California, including two streams entering San Francisco Bay: Arroyo Corte Madera Del Presidio and Corte Madera Creek. This critical habitat designation includes all waterways, substrate, and adjacent riparian zones of estuarine and riverine reaches (including off-channel habitats) below longstanding naturally impassable barriers (i.e. natural waterfalls in existence for at least several hundred years). These areas are important for the species' overall conservation by protecting growth, reproduction, and feeding.

Lower Columbia River coho salmon

Distribution. The lower Columbia River Coho salmon ESU includes all naturally spawned populations of Coho salmon in the Columbia River and its tributaries in Washington and Oregon, from the mouth of the Columbia up to and including the Big White Salmon and Hood Rivers, and includes the Willamette River to Willamette Falls, Oregon (Figure 14). Twenty-five artificial propagation programs are part of this ESU.

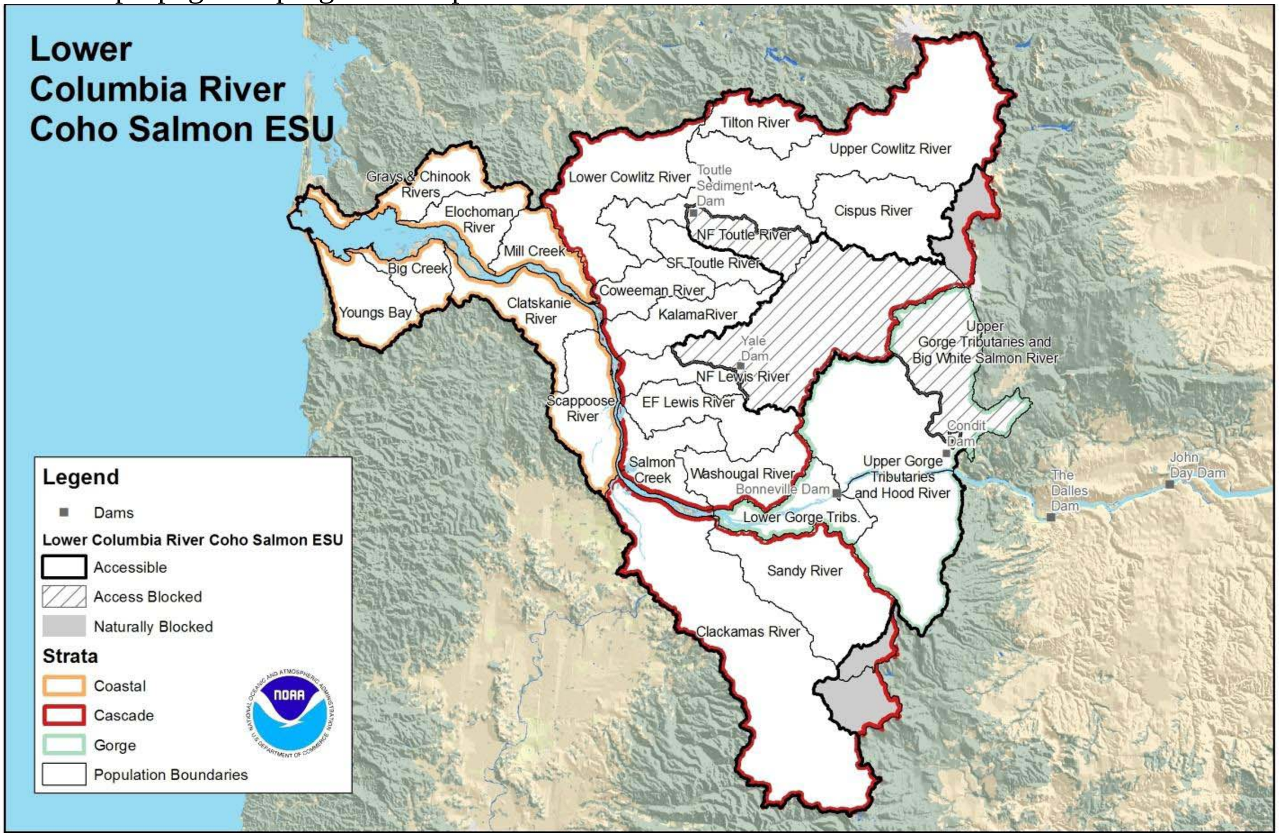

Figure 14. Population boundaries, dams, accessible areas, and extirpated reaches of lower Columbia River coho distribution, incorporating critical habitat boundaries (dark black line). Taken from NMFS (2011).

Two distinct runs distinguished by the timing of adult returns to freshwater (early returners and later returners) occur within the ESU. Early returning adults generally migrate south of the Columbia River once they reach the ocean, returning to freshwater in mid-August and to spawning tributaries in early September. Peak spawning of early returning adults occurs from mid-October to early November. Late returning adults exhibit a northern oceanic distribution, returning to the Columbia River from late September through December, and enter tributaries from October through January. Most late return adults spawn between November through January, although some spawn in February and as late as March (LCFRB 2004). Almost all Lower Columbia River ESU Coho salmon females and most males spawn at 3 years of age.

Status and trends. The NMFS listed Lower Columbia River Coho salmon as endangered on June 28, 2005 (70 FR 37160). Twenty four historical populations occur in the ESU; 21 are at high risk of extinction while the other three are fairing somewhat better (NMFS 2011d). The vast majority (over 90%) of the historic population in the Lower Columbia River Coho salmon ESU appear to be either extirpated or nearly so. Recent counts of natural-origin spawners and the recent fraction of hatchery-origin spawners are noted in Table 16, where available.

Table 16. Lower Columbia River Coho salmon populations and selected measures of population viability.

River	**2002 spawner count**[a]	**Geometric mean abundance 2000-2002**[b]	**Percent hatchery contributions**[c]	**Long-term median growth rate (λ)**[d]
Youngs Bay and Big Creek	4,473		91	
Grays River				
Elochoman River				
Clatskanie River	229		60	
Mill, Germany, and Abernathy creeks				
Scappoose Rivers	458		0	
Cispus River				
Tilton River				
Upper Cowlitz River				
Lower Cowlitz River				
North Fork Toutle River				
South Fork Toutle River				
Coweeman River				
Kalama River				
North Fork Lewis River				
East Fork Lewis River				
Upper Clackamas River	1,001	2,122	12	1.009 (0.898-1.177)
Lower Clackamas River	2,402		78	
Salmon Creek				
Upper Sandy River	310	643	0	1.012 (0.874-1.172)
Lower Sandy River	271		97	
Washougal River				
Columbia River Gorge – lower tributaries				
White Salmon				
Columbia River Gorge – upper tributaries	1,317		>65	
Hood River				

[a] All data are reported in Good *et al.* 2005. Spawner data from 2002 only.
[b] Geometric mean number of Coho salmon above the dams. * is a combined total for the upper and lower Clackamas River. Reported in Good *et al.* 2005
[c] Hatchery production likely dominates yearly returns for the ESU as a whole.
[d] The λ calculated estimates the natural growth rate after accounting for hatchery-origin spawners. The estimate provided above assumes that hatchery-origin spawners make no reproductive contribution. The λ for the Clackamas

River is calculated with data spanning 1973-2002, and for the Sandy River covers 1977-2002. The Clackamas River value includes both early-run and late-run Coho salmon.

Only two populations of Coho salmon within this ESU produce a sizeable number of naturally spawned fish, the upper Sandy River population above Marmot Dam and the Clackamas River population above the North Fork Dam. Excluding these, natural spawner abundance is below 500 individuals (NMFS 2011d). A survey from Mill/Germany/Abernathy creeks produced an estimate of 3,150 spawners in 2006; over half of these were hatchery fish (NMFS 2011d). The long-term and short-term trends for Marmot Dam counts are both negative. The long-term median growth rate is slightly positive for both the Sandy and Clackamas rivers, but the confidence intervals for each are very wide indicating there is a large amount of uncertainty. Both populations within the Sandy and Clackamas rivers have suffered from recruitment failure a number of times over the past 15 years.

The most serious threat facing this ESU is the scarcity of naturally-produced spawners, with attendant risks associated with small population, loss of diversity, and fragmentation and isolation of the remaining naturally-produced fish. Spatial structure has been substantially reduced by the loss of access to upper basins from tributary hydro development (i.e., Condit Dam on the Big White Salmon River and Powerdale Dam on the Hood River). The diversity of populations in all three areas has been eroded by large hatchery influences and periodically, low effective population sizes.

Critical habitat. The NMFS has not designated critical habitat for Lower Columbia River Coho salmon.

Species-specific threats and limitations to recovery. Several direct and habitat-related factors inhibit the recovery of Lower Columbia River Coho salmon (LCFRB 2010, NMFS 2011d). Habitat impacts include degraded estuarine and near-shore marine habitat resulting from cumulative impacts of land use and flow management by the Columbia River hydropower system, fish passage barriers that limit access to spawning and rearing habitats, floodplain connectivity and function, channel structure and complexity, riparian areas and large wood supply, stream substrate, stream flow, and water quality have been degraded as a result of cumulative impacts of agriculture, forestry, and development. Additional threats to habitat include an altered flow regime and Columbia River plume has altered the temperature regime and estuarine food web, and has reduced ocean productivity, reduced access to off-channel rearing habitat in the lower Columbia River, as well as reduced productivity resulting from sediment and nutrient-related changes in the estuary. Other impacts also include hatchery- and harvest-related effects, juvenile fish strandings that result from ship wakes, and contaminants affecting fish health and reproduction.

Oregon Coast coho salmon

Distribution. The Oregon Coast Coho salmon ESU includes all naturally spawned populations of Coho salmon in Oregon coastal streams south of the Columbia River and north of Cape Blanco (63 FR 42587; August 1998). One hatchery population, the Cow Creek hatchery Coho salmon, is considered part of the ESU. Table 17 identifies populations within the Oregon Coast Coho salmon ESU, their abundances, and hatchery input.

Table 17. Oregon Coast Coho populations and selected measures of population viability.

Basin[a]	Mean Spawner Abundance[b]	13-Year Spawner Trend (SE)[c]	Percent Hatchery Contribution[d]
Necanicum	1,889	1.169 (0.860)	2.9-6.4
Nehalem	18,741	1.206 (0.889)	0.5-26.0
Tillamook Bay	3,949	1.191 (1.084)	0-5.6
Nestucca	3,846	1.230 (1.015)	0-10.4
Siletz	2,295	1.070 (0.760)	1.8-100
Yaquima	3,665	1.204 (1.205)	0-37.5
Alsea	3,621	1.042 (0.960)	0-87.5
Siuslaw	16,213	1.120 (1.037)	0.3-11.1
Umpqua	24,351	1.182 (0.662)	2.1-8.3
Coos	20,136	1.088 (1.066)	0-1.9
Coquille	8,847	1.070 (0.649)	0-6.0

[a] Population structure is unclear. The above data reflects the assumption that spawners from major river basins are largely isolated, and each basin comprises a population. All data are reported in Good *et al.* 2005.
[b] Recent 3-year geometric mean of natural-origin spawners.
[c] Data years 1990-2002.
[d] Data represents the range of percent hatchery contributions from 1998 through 2002 (from Jacobs *et al.* 2002, 2001, and 2002 as cited in Good *et al.* 2005).

Status and trends. The Oregon coast Coho salmon ESU was listed as a threatened species under the ESA on February 11, 2008 (73 FR 7816), the conclusion to a 13-year history of court cases. The most recent NMFS status review for the Oregon Coast Coho ESU was conducted by the biological review team in 2003, which assessed data through 2002. The abundance and productivity of Oregon Coast Coho since the previous status review represented some of the best and worst years on record (NMFS 1997a). Yearly adult returns for the Oregon Coast Coho ESU were over 160,000 natural spawners in 2001 and over 260,000 in 2002, far exceeding the abundance observed for the past several decades. These increases in spawner abundance in 2000 to 2002 followed three consecutive brood years (the 1994-1996 brood years returning in 1997 to 1999, respectively) exhibiting recruitment failure (recruitment failure is when a given year class of natural spawners fails to replace itself when its offspring return to the spawning grounds 3 years later). These 3 years of recruitment failure were the only such instances observed thus far in the entire 55-year abundance time series for Oregon Coast Coho salmon (although comprehensive population-level survey data have only been available since 1980). The 2000 to 2002 increases in natural spawner abundance occurred in many populations in the northern portion of the ESU, which were the most depressed at the time of the last review (NMFS 1997a). Although encouraged by the increase in spawner abundance in 2000-2002, the biological review team noted that the long-term trends in ESU productivity were still negative due to the low abundances observed during the 1990s.

Since the biological review team convened, the total abundance of natural spawners in the Oregon Coast Coho ESU has declined each year (i.e., 2003 to 2006). The abundance of total natural spawners in 2006 (111,025 spawners) was approximately 43% of the recent peak abundance in 2002 (255,372 spawners). In 2003, ESU-level productivity (evaluated in terms of the number of spawning recruits resulting from spawners 3 years earlier) was above replacement, and in 2004, productivity was approximately at replacement level. However, productivity was below replacement in 2005 and 2006, and dropped to the lowest level since 1991 in 2006 (73 FR 7816).

Preliminary spawner survey data for 2007 (the average peak number of spawners per mile observed during random Coho spawning surveys in 41 streams) suggest that the 2007 to 2008 return of Oregon Coast Coho is either (1) much reduced from abundance levels in 2006, or (2) exhibiting delayed run timing from previous years. As of December 13, 2007, the average peak number of spawners per mile was below 2006 levels in 38 of 41 surveyed streams (ODFW 2007 *in* 73 FR 7816). It is possible that the timing of peak spawner abundance is delayed relative to previous years, and that increased spawner abundance in late December and January 2008 will compensate for the low levels observed thus far.

The recent 5-year geometric mean abundance (2002 to 2006) of approximately 152,960 total natural spawners remains well above that of a decade ago (approximately 52,845 from 1992 to 1996). However, the decline in productivity from 2003 to 2006, despite generally favorable marine survival conditions and low harvest rates, is of concern (73 FR 7816).

Critical habitat. The NMFS designated critical habitat for Oregon Coast Coho on February 11, 2008 (73 FR 7816). The designation includes 72 of 80 watersheds occupied by Oregon Coast Coho salmon, and totals about 6,600 stream miles including all or portions of the Nehalem, Nestucca/Trask, Yaguina, Alsea, Umpqua and Coquille basins. These areas are essential for feeding, migration, spawning, and rearing. The specific primary constituent elements include: 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; freshwater migratory corridors free of obstruction with adequate water quantity and quality conditions; and estuarine, nearshore and offshore areas free of obstruction with adequate water quantity, quality and salinity conditions that support physiological transitions between fresh- and saltwater, predator avoidance, foraging and other life history behaviors.

The historical disturbance regime in the central Oregon Coast Range was dominated by a mixture of high and low-severity fires, with a natural rotation of approximately 271 years. Old-growth forest coverage in the Oregon Coast Range varied from 25 to 75% during the past 3,000 years, with a mean of 47%, and never fell below 5% (Wimberly *et al.* 2000). Currently, the Coast Range has approximately 5% old-growth, almost all of it on Federal lands. The dominant disturbance now is logging on a cycle of approximately 30 to 100 years, with fires suppressed.

The State of Oregon (2005) completed an assessment of habitat conditions in the range of Oregon coast coho salmon in 2005. Oregon's assessment mapped how streams with high intrinsic potential for coho salmon rearing are distributed by land ownership categories. Agricultural lands and private industrial forests have by far the highest percentage of land ownership in high intrinsic potential areas and along all coho salmon stream miles. Federal lands have only about 20% of coho salmon stream miles and 10% of high intrinsic potential stream reaches. Because of this distribution, activities in lowland agricultural areas are particularly important to the conservation of Oregon coast coho salmon.

The Oregon coast coho salmon assessment concluded that at the scale of the entire domain, pools are generally abundant, although slow-water and off-channel habitat (which are important refugia for coho salmon during high winter flows) are limited in the majority of streams when compared to reference streams in minimally-disturbed areas. Amounts of large wood in streams are low in all four Oregon Department of Fish and Wildlife monitoring areas and land-use types relative to reference conditions. Amounts of fine sediment are high in three of the four monitoring areas, and were comparable to reference conditions only on public lands. Approximately 62 to 91% of tidal wetland acres (depending on estimation procedures) have been lost for functionally and potentially independent populations of coho salmon.

As part of the coastal coho salmon assessment, the Oregon Department of Environmental Quality analyzed the status and trends of water quality in the range of Oregon coast coho salmon using the Oregon water quality index, which is based on a combination of temperature, dissolved oxygen, biological oxygen demand, pH, total solids, nitrogen, total phosphates, and bacteria. Using the index at the species scale, 42% of monitored sites had excellent to good water quality, and 29% show poor to very poor water quality. Within the four monitoring areas, the North Coast had the best overall conditions (six sites in excellent or good condition out of nine sites), and the Mid-South coast had the poorest conditions (no excellent condition sites, and only two out of eight sites in good condition). For the 10-year period monitored between 1992 and 2002, no sites showed a declining trend in water quality. The area with the most improving trends was the North Coast, where 66% of the sites (six out of nine) had a significant improvement in index scores. The Umpqua River basin, with one out of nine sites (11%) showing an improving trend, had the lowest number of improving sites.

Species-specific threats and limitations to recovery. Oregon coast coho salmon are impaired in their recovery by several anthropogenic factors, including floodplain connectivity and function, channel structure and complexity, riparian areas and large wood supply, stream substrate, stream flow, and water quality that have been degraded as a result of cumulative impacts of agriculture, forestry, instream mining, dams, road crossings, dikes, levees, etc. (Stout *et al.* 2011). In addition, fish passage barriers limit access to spawning and rearing habitats and adverse climate, altered past ocean/marine productivity, and current ocean ecosystem conditions have favored competitors and predators and reduced salmon survival rates in freshwater rivers and lakes, estuaries, and marine environments

Southern Oregon/Northern California coast coho salmon

Distribution. Southern Oregon/Northern California coast coho salmon consists of all naturally spawning populations of coho salmon that reside below long-term, naturally impassible barriers in streams between Punta Gorda, California and Cape Blanco, Oregon, as well as three artificial propagation programs: the Cole Rivers Hatchery, Trinity River Hatchery, and Iron Gate Hatchery coho hatchery programs. The three major river systems supporting Southern Oregon – Northern Coastal California coast coho are the Rogue, Klamath (including the Trinity), and Eel rivers.

Southern Oregon and Northern California coast coho immigrate to natal rivers in September or October. River entry is much later south of the Klamath River Basin, occurring in November and December, as well as in basins south of the Klamath River to the Mattole River, California. River entry occurs from mid-December to mid-February in rivers farther south. Because individuals enter rivers late, they spend much less time in the river. Coho salmon adults spawn at age 3, spending just over 1 year in fresh water and a year and a half in the ocean.

Status and trends. Southern Oregon/Northern California coast coho salmon were listed as threatened on May 7, 1997 (62 FR 24588); they retained that classification when their status was reviewed on June 28, 2005 (70 FR 37160). Southern Oregon/Northern California Coast coho salmon extend from Cape Blanco in southern Oregon to Punta Gorda in northern California (Weitkamp *et al.* 1995b). The status of coho salmon coast-wide, including the Southern Oregon/Northern California Coast coho salmon ESU, was formally assessed in 1995 (Weitkamp *et al.* 1995b). Two subsequent status review updates have been published by NMFS, one addressing all West Coast coho salmon ESUs and a second specifically addressing the Oregon Coast Southern Oregon/Northern California Coast coho salmon ESUs (NMFS 1996, 1997a). In the 1997 status update, estimates of natural population abundance were based on very limited information. New data on presence/absence in northern California streams that historically supported coho salmon were even more disturbing than earlier results, indicating that a smaller percentage of streams contained coho salmon compared to the percentage presence in an earlier study. However, it was unclear whether these new data represented actual trends in local extinctions or were biased by sampling effort.

Data on population abundance and trends are limited for the California portion of this ESU. No regular estimates of natural spawner escapement are available. Historical point estimates of coho salmon abundance for the early 1960s and mid-1980s suggest that statewide coho spawning escapement in the 1940s ranged between 200,000 and 500,000 fish. Numbers declined to about 100,000 fish by the mid-1960s with about 43% originating from this ESU. Brown *et al.* (1994) estimated that the California portion of this ESU was represented by about 7,000 wild and naturalized coho salmon (Good *et al.* 2005b). In the Klamath River, the estimated escapement has dropped from approximately 15,400 in the mid-1960s to about 3,000 in the mid 1980s, and more recently to about 2,000 (Good *et al.* 2005b). The second largest producing river in this ESU, the Eel River, dropped from 14,000, to 4,000 to about 2,000 during the same period. Historical estimates are considered “best guesses” made using a combination of limited catch statistics, hatchery records, and the personal observations of biologists and managers. Although quantitiative estimates of abundance in this ESU are rare, the best indications are that since 2005, populations within the ESU have continued to decline (NMFS 2011h).

Most recently, Williams *et al.* (2006) described the structure of historic populations of Southern Oregon/Northern California Coast coho salmon. They described three categories of populations: functionally independent populations, potentially independent populations and dependent populations. Functionally independent populations are populations capable of existing in isolation with a minimal risk of extinction. Potentially independent populations are similar but rely on some interchange with adjacent populations to maintain a low probability of extinction. Dependent populations have a high risk of extinction in isolation over a 100-year timeframe and rely on exchange of individuals from adjacent populations to maintain themselves.

Critical habitat. NMFS designated critical habitat for Southern Oregon/Northern California Coast coho salmon on May 5, 1999 (64 FR 24049). Critical habitat for this species encompasses all accessible river reaches between Cape Blanco, Oregon, and Punta Gorda, California. Critical habitat consists of the water, substrate, and river reaches (including off-channel habitats) in

specified areas. Accessible reaches are those within the historical range of the ESU that can still be occupied by any life stage of coho salmon. Of 155 historical streams for which data are available, 63% likely still support coho salmon. These river habitats are important for a variety of reasons, such as supporting the feeding and growth of juveniles and serving as spawning habitat for adults. Limiting factors identified for this species include: loss of channel complexity, connectivity and sinuosity, loss of floodplain and estuarine habitats, loss of riparian habitats and large in-river wood, reduced stream flow, poor water quality, temperature and excessive sedimentation, and unscreened diversions and fish passage structures.
The Elk River flows through Curry County, and drains approximately 92 square miles (or 58,678 acres)(Maguire 2001). Historical logging, mining, and road building have degraded stream and riparian habitats in the Elk River basin. Limiting factors identified for salmon and steelhead production in this basin include sparse riparian cover, especially in the lower reaches, excessive fine sediment, high water temperatures, and noxious weed invasions (Maguire 2001).

The Rogue River drains approximately 5,160 square miles within Curry, Jackson and Josephine counties in southwest Oregon. The mainstem is about 200 miles long and traverses the coastal mountain range into the Cascades. The Rogue River estuary has been modified from its historical condition. Jetties were built by the Army Corps of Engineers in 1960, which stabilized and deepened the mouth of the river. A dike that extends from the south shore near Highway 101 to the south jetty was completed in 1973. This dike created a backwater for the large shallow area that existed here, which has been developed into a boat basin and marina, eliminating most of the tidal marsh.

The quantity of estuary habitat is naturally limited in the Rogue River. The Rogue River has a drainage area of 5,160 square miles, but the estuary at 1,880 acres is one of the smallest in Oregon. Between 1960 and 1972, approximately 13 acres of intertidal and 14 acres of subtidal land were filled in to build the boat basin dike, the marina, north shore riprap and the other north shore developments (Hicks 2005). Jetties constructed in 1960 to stabilize the mouth of the river and prevent shoaling have altered the Rogue River, which historically formed a sill during summer months (Hicks 2005).

The Lower Rogue Watershed Council's watershed analysis (Hicks 2005) lists factors limiting fish production in tributaries to Lower Rogue River watershed. The list includes water temperatures, low stream flows, riparian forest conditions, fish passage and over-wintering habitat. Limiting factors identified for the Upper Rogue River basin include fish passage barriers, high water temperatures, insufficient water quantity, lack of large wood, low habitat complexity, and excessive fine sediment (RBCC 2006).

The Chetco River estuary has been significantly modified from its historical condition. Jetties were erected by the Army Corps of Engineers in 1957, which stabilized and deepened the mouth of the river. These jetties have greatly altered the mouth of the Chetco River and how the estuary functions as habitat for salmon migrating to the ocean. A boat basin and marina were built in the late 1950s and eliminated most of the functional tidal marsh. The structures eliminated shallow water habitats and vegetation in favor of banks stabilized with riprap. Since then, nearly all remaining bank habitat in the estuary has been stabilized with riprap. The factors limiting fish production in the Chetco River appear to be high water temperature caused by lack of shade, especially in tributaries, high rates of sedimentation due to roads, poor over-wintering habitat due to a lack of large wood in tributaries and the mainstem, and poor quality estuary habitat (Maguire 2001).

Species-specific threats and limitations to recovery. Numerous anthropogenic factors hinder recovery of Southern Oregon/Northern California Coast coho salmon (NMFS 2007c, 2011h). These include lack of floodplain and channel structure, impaired water quality, estuarine function, fish passage, adverse hatchery-related effects, predation, competition, and disease, altered sediment transport, degraded stream substrate, altered hydrologic function due to altered amount and timing of river flows, and degraded riparian forest conditions and large wood recruitment.

Sockeye salmon

Description of the species. Sockeye salmon are the second most abundant of the seven Pacific salmon species, and occur in the North Pacific and Arctic oceans and associated freshwater systems. This species ranges south as far as the Sacramento River in California and northern Hokkaido in Japan, to as far north as far as Bathurst Inlet in the Canadian Arctic and the Anadyr River in Siberia (Burgner 1991). The largest populations, and hence the most important commercial populations, occur north of the Columbia River.

Sockeye salmon exhibit a very diverse life history, characteristically using both riverine and lake habitat, exhibiting both freshwater resident and anadromous forms. The vast majority of sockeye salmon are anadromous fish that make use of lacustrine habitat for juvenile rearing. These “lake-type” fish typically spawn in the outlet streams of lakes and occasionally in the lakes themselves. Juvenile sockeye salmon will then use the lake environment for rearing for up to 3 years before migrating to sea. After 1 to 4 years at sea, sockeye salmon will return to their natal lake to spawn. Some sockeye, however, spawn in rivers without lake habitat for juvenile rearing. Offspring of these riverine spawners tend to use the lower velocity sections of rivers as the juvenile rearing environment for 1 to 2 years, or may migrate to sea in their first year.

Sockeye salmon also have a wholly freshwater life history form, called kokanee (Burgner 1991). In some cases a single population will give rise to both the anadromous and freshwater life history form. While in freshwater juveniles of both life history types prey primarily upon insects. The presence of both life history types may be related to the energetic costs of outmigrating to sea, and the productivity of the lacustrine system they inhabit. In coastal lakes, where the migration to sea is relatively short and energetic costs are minimal, kokanee populations are rare.

Habitat. Once smolts enter the Pacific Ocean, they distribute widely across the North Pacific, generally above 40º N, where a current boundary is located. Season, temperature, salinity, life stage, age, size, availability of prey and population-of-origin are all factors that influence offshore movements (Burgner 1991). Sockeye tend to occupy the upper water column, within 30 m of the surface and preferably within 15 m and tend to be closer to the surface at night versus daytime (Manzer 1964, French *et al.* 1976, Pauley *et al.* 1989, Burgner 1991). Sockeye migrate several thousand miles in search of prey and are considered to travel continuously (Royce *et al.* 1968). Juveniles leaving freshwater move north along the Alaskan coastal belt beginning in mid-

May until early winter when they disperse into the Gulf of Alaska (Emmett *et al.* 1991b, NMFS-AKR 2005).

Thermoclines may also influence vertical distribution, with fish only mingling between surface and deeper waters when the boundary temperature difference is weak. Sockeye appear to prefer cooler waters relative to other salmon species, but younger salmon may prefer warmer sea surface temperatures (optimally 15° C) than larger, older fish (12° and 14° C), possibly an artifact of older fish being distributed further north (Pauley *et al.* 1989). Adult upstream migration may be blocked by temperatures above 21º C (McCullough 1999). However, temperatures below 21º C can stress fish by increasing their susceptibility to disease and elevating their metabolism (Brett 1979). Maturation and timing of return to spawn by sockeye appears to be linked to water temperature, with gonad development increasing in late May through early July (Nishiymama 1984).

Feeding. While in freshwater, juveniles prey primarily upon insects. While at sea, sockeye prey upon a variety of organisms, including small fish (capelin, lantern fish, cod, sand lance, herring, and pollock), squid, crustacean larvae, krill, and other invertebrates (Foerster 1968, French *et al.* 1976, Wing 1977).

Reproduction. Spawning generally occurs in late summer and autumn, but the precise time can vary greatly among populations. Age at maturity varies by region from 2 to 5 years, but is generally 2 to 4 years in Washington State (Burgner 1991). Males often arrive earlier than females on the spawning grounds, and will persist longer during the spawning period. Average fecundity ranges from about 2,000-2,400 eggs per female up to 5,000 eggs, depending upon the population and average age of the female. Fecundity in kokanee is much lower and may range from about 300-2,000 eggs.

Incubation is a function of water temperatures, but generally lasts between 100 and 200 days (Burgner 1991). After emergence, fry move rapidly downstream or upstream along the banks to the lake rearing area. Fry emerging from lakeshore or island spawning grounds may simply move along the shoreline of the lake (Burgner 1991).

Threats. 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 2011l). In the ocean, marine mammals and other fish prey on sockeye, 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 2011l).

Ozette Lake sockeye salmon

Distribution. This ESU includes all naturally spawned sockeye salmon in Ozette Lake, Ozette River, Coal Creek, and other tributaries flowing into Ozette Lake, Washington (Figure 15). Composed of only one population, the Ozette Lake sockeye salmon ESU consists of five spawning aggregations or subpopulations which are grouped according to their spawning locations. The five spawning locations are Umbrella and Crooked creeks, Big River, and

Olsen’s and Allen’s beaches (NMFS 2009c).

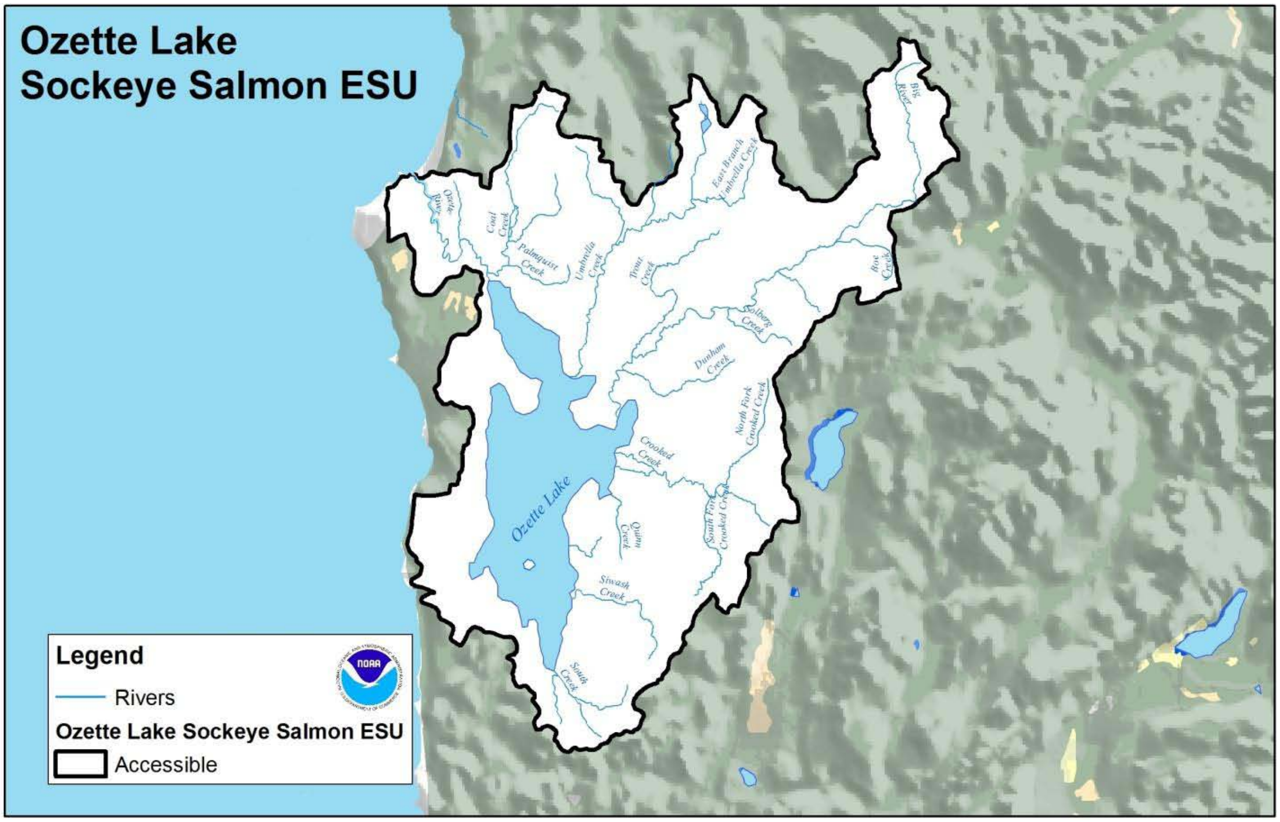

Figure 15. Population boundaries of Lake Ozette sockeye distribution, incorporating critical habitat boundaries (dark black line). Taken from NMFS (2011).

Adult Ozette Lake sockeye salmon enter Ozette Lake through the Ozette River from mid-April to mid-August, holding three to nine months in Ozette Lake prior to spawning in late October through January. Sockeye salmon spawn primarily in lakeshore upwelling areas in Ozette Lake (particularly at Allen's Bay and Olsen's Beach), and in two tributaries Umbrella Creek and Big River. Minor spawning may occur below Ozette Lake in the Ozette River or in Coal Creek, a tributary of the Ozette River. Beach spawners are almost all age-4 adults, while tributary spawners are ages 3 and 5 (NMFS 2009c). Spawning occurs in the fall through early winter, with peak spawning in tributaries in November and December. Eggs and alevins remain in the gravel until the fish emerge as fry in spring. Fry then migrate immediately to the limnetic zone in Ozette Lake, where the fish rear. After one year of rearing, in late spring, Ozette Lake sockeye salmon emigrate seaward as age-1+ smolts, where they spend between 1 and 3 years in ocean before returning to freshwater.

Status and trends. The NMFS originally listed Ozette Lake sockeye salmon ESU as a threatened species in 1999 (64 FR 14528). This classification was retained on June 28, 2005 (70 FR 37160). This ESU includes all naturally spawned populations of sockeye salmon in Ozette Lake, Ozette River, Coal Creek, and other tributaries flowing into Ozette Lake, Washington. Two artificial propagation programs are considered part of this ESU. The NMFS considers these artificially propagated populations no more divergent relative to the local natural population than would be expected between closely related natural populations (70 FR 37160).

The historical abundance of Ozette Lake sockeye salmon is poorly documented, but may have been as high as 50,000 individuals (Blum 1988). The overall abundance of naturally–produced Ozette Lake sockeye salmon is believed to have declined substantially from historical levels. In the first study of lake escapement of Ozette Lake sockeye salmon (Kemmerich 1945), the run size entering the lake was estimated at a level of several thousand fish. These counts appear to be roughly double the current mean lake abundance, considering that they were likely conducted upstream from fisheries in or near to the Ozette River. Makah Fisheries Management (MFM 2000) concluded that there appears to be a substantial decline in the tribal catch of Ozette Lake sockeye salmon beginning in the 1950s and a similar decline in the run size since the 1920s weir counts reported by Kemmerich (1945).

An analysis of total annual Ozette Lake sockeye salmon abundance (based on adult run size data presented in Jacobs *et al.* 1996) indicates a trend in abundance averaging -2% per year over the period 1977 through 1998 (NMFS 1998c). The current tributary-based hatchery program was planned and initiated in response to the declining population trend identified for the Ozette Lake sockeye salmon population. The most recent (1996 to 2003) run-size estimates range from a low of 1,609 in 1997 to a high of 5,075 in 2003, averaging approximately 3,600 sockeye per year (NMFS 2009c). For return years 2000 to 2003, the 4-year average abundance estimate was slightly over 4,600 sockeye. Because run-size estimates before 1998 are likely to be even more unreliable than recent counts, and new counting technology has resulted in an increase in estimated run sizes; no statistical estimation of trends is reported. The current trends in abundance are unknown for the beach spawning aggregations. Although overall abundance appears to have declined from historical levels, whether this resulted in fewer spawning aggregations, lower abundances at each aggregation, or both, is not known (Good *et al.* 2005b). Based on estimates of habitat carrying capacity, a viable sockeye salmon population in Lake Ozette watershed would range between 35,500 to 121,000 spawners (Rawson *et al.* 2009).

There has been no harvest of Ozette Lake sockeye salmon for the past four brood-cycle years (since 1982). Prior to that time, ceremonial and subsistence harvests by the Makah Tribe were low, ranging from 0 to 84 fish per year. Harvest has not been an important mortality factor for the population in over 35 years. In addition, due to the early river entry timing of returning Ozette Lake sockeye salmon (beginning in late April, with the peak returns prior to late-May to mid-June), the fish are not intercepted in Canadian and United States marine area fisheries directed at Fraser River sockeye salmon. There are currently no known marine area harvest impacts on Ozette Lake sockeye salmon.

Overall abundance is substantially below historical levels (Good *et al.* 2005b). Declines in abundance have been attributed to a combination of introduced species, predation, loss of tributary populations, a loss of quality of beach spawning habitat, temporarily unfavorable ocean conditions, habitat degradation, and excessive historical harvests (Jacobs *et al.* 1996). In the last few years the number of returning adults has increased, although some of these individuals are of hatchery origin. This produces uncertainty regarding natural growth rate and productivity of the ESU's natural component. In addition, genetic integrity has perhaps been compromised due to the artificial supplementation that has occurred in this population, since approximately one million sockeye have been released into the Ozette watershed from the late 1930s to present

(Kemmerich 1945, Boomer 1995).

Critical habitat. On September 2, 2005, the NMFS designated critical habitat for the Ozette Lake sockeye salmon ESU (70 FR 52630). The specific geographic areas designated as critical are the Hoh/Quillayute Subbasin, Ozette Lake and the Ozette Lake watershed, and the Ozette River upstream to endpoints in Big River, Coal Creek, East Branch Umbrella Creek, the North and South Fork of Crooked Creek and several other tributaries. The specific primary constituent elements identified for Lake Ozette sockeye 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, and adequate passage conditions. Only one watershed supports this ESU, and it is rated as having a high conservation value. This watershed is essential to the species' overall conservation by protecting quality growth, reproduction, and feeding.

The Ozette Lake tributary basin is 77 mi^2 and includes several large tributaries and numerous smaller tributaries. Currently, land ownership in the watershed is 73% private land, 15% Olympic National Park, 11% Washington State, and 1% Tribal. Natural disturbance in the watershed was dominated by wind and hydrogeomorphic events, while contemporary disturbance additionally includes logging, road construction and maintenance, residential and agricultural development, stream channelization and direct and indirect stream wood clearance. These activities alter stream flow patterns and elevate of sediment loads and increased sedimentation within drainage basins. Wood removal has resulted in less hydraulic roughness, reduced instream water depths, and reduced backwater effects on Lake Ozette, which has thus altered the entire hydraulic control on Lake Ozette levels and changed the in-river stage-discharge relationship. More recently, deposition of sediment originating from Coal Creek at the lake outlet has further altered lake and river levels (Haggerty *et al.* 2009b).

Private timber companies own approximately 93% of the four largest tributary watersheds to Lake Ozette. Logging accelerated over the period of record, with 8.7% of the entire Ozette Lake basin clear-cut by 1953, increasing to 83.6% of the basin area clear-cut by 2003 (Haggerty *et al.* 2009b). Effects associated with logging depended on stream size, gradient, and time after harvest. In high-energy coast streams, landslides and debris torrents often modify steep slope tributaries and the mainstem of creeks. Bank erosion also alters the stream channel on the alluvial flood plain. These effects are additive in the system and reduced the quality of spawning and rearing habitat for juvenile salmonids (Hartman *et al.* 1996). Lower gradient streams typically will have an accumulation of sediment. Second-growth logged sections (1 2- 35 yr after logging), re-shaded by deciduous forest canopy, have lower biomass of trout and fewer predator taxa than old-growth sites (Murphy and Hall 1981). Based on the quantity and quality of the physical and biological features, the conservation value of the Ozette Lake HUC5 watershed (#1710010102) for sockeye salmon is likely to be "high" (NMFS 2005e).

Species-specific threats and limitations to recovery. Floodplain connectivity and function, channel structure and complexity, riparian areas and large wood supply, lake beach spawning habitat, and stream substrate have been degraded as a result of cumulative impacts of forest practices, agriculture, and development (Haggerty *et al.* 2009a, USDC 2009a, Muir and Williams

2011). Harbor seals and river otters, and predaceous non-native and native fish species, are reducing the abundance of adult fish that successfully spawn, and the abundance of sockeye smolts escaping seaward from the watershed each year.

Snake River sockeye salmon

Distribution. Snake River sockeye salmon are unique compared to other sockeye populations: they spawn at a higher elevation (6,500 feet) and have a longer freshwater migration (approximately 900 miles) than any other sockeye salmon population (Figure 16). Sockeye salmon in this ESU spawn in Redfish Lake in Idaho's Stanley Basin (Bjornn *et al.* 1968, Foerster 1968). Stanley Basin sockeye salmon are separated by 700 or more river miles from two other extant upper Columbia River populations in the Wenatchee River and Okanogan River drainages. These latter populations return to lakes at substantially lower elevations (Wenatchee at 1,870 feet and Okanagon at 912 feet) and occupy different ecoregions. The Snake River sockeye salmon ESU includes all anadromous and residual sockeye salmon from the Snake River basin of Idaho, as well as hatchery individuals from the Redfish Lake Captive Broodstock Program.

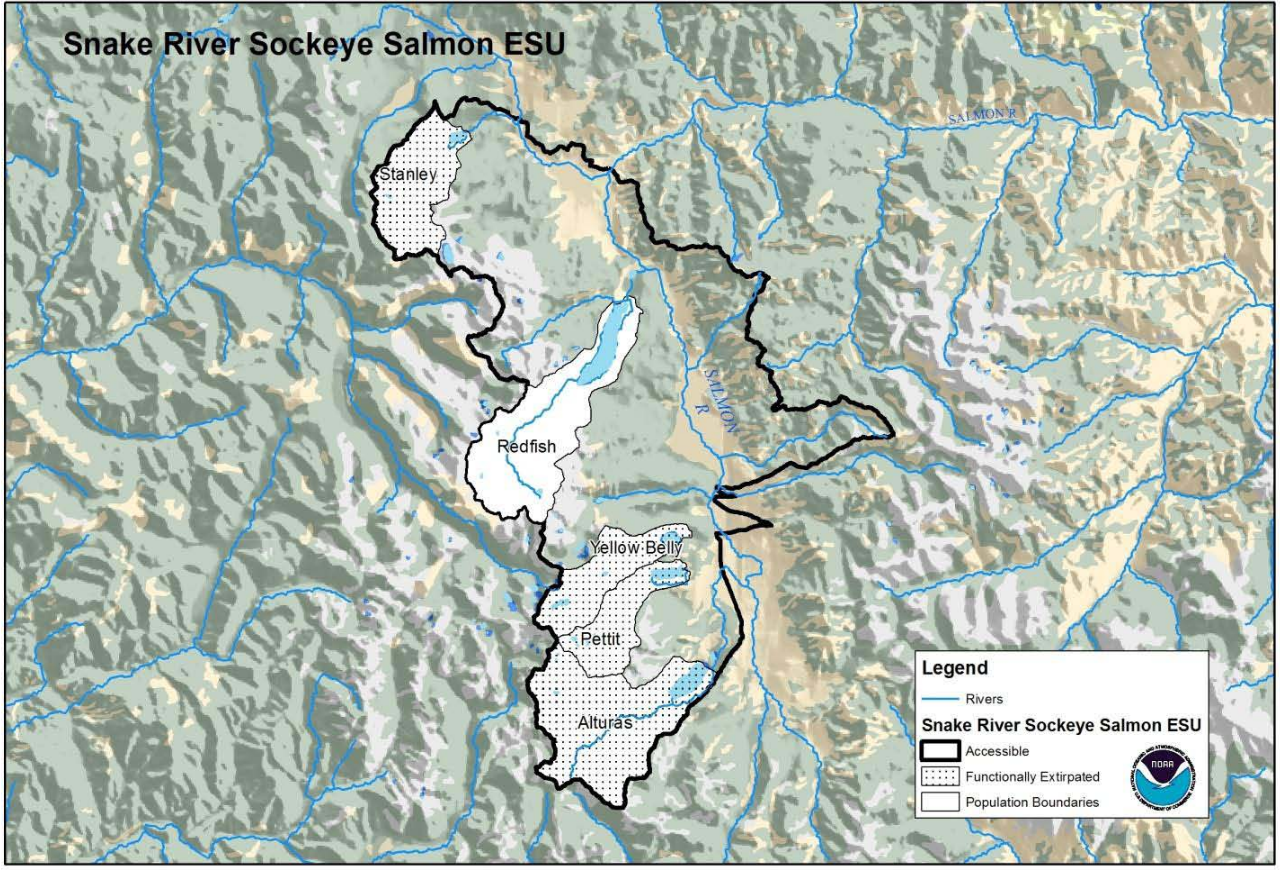

Figure 16. Population boundaries, accessible areas, and extirpated reaches of Snake River sockeye distribution, incorporating critical habitat boundaries (dark black line). Taken from NMFS (2011l).

Status and trends. Snake River sockeye salmon were originally listed as endangered in 1991 and the NMFS retained that classification when their status was reviewed on June 28, 2005 (70 FR 37160). The only extant sockeye salmon population in the Snake River basin at the time of listing was that in Redfish Lake. Other lakes in the Snake River basin historically supported sockeye salmon populations, including Wallowa Lake (Grande Ronde River drainage, Oregon), Payette Lake (Payette River drainage, Idaho) and Warm Lake (South Fork Salmon River drainage, Idaho (Waples *et al.* 1997). These populations are now considered extinct. Although kokanee, a resident form of *O. nerka*, occur in numerous lakes in the Snake River basin, other lakes in the Stanley Basin, and sympatrically with sockeye in Redfish Lake, resident *O. nerka* were not considered part of the species at the time of listing (1991). Subsequent to the 1991 listing, a residual form of sockeye residing in Redfish Lake was identified. The residuals are non-anadromous, completing their entire life cycle in freshwater, but spawn at the same time and in the same location as anadromous sockeye salmon ESU. In 1993, the NMFS determined that residual sockeye salmon in Redfish Lake were part of the Snake River sockeye salmon. Also, artificially propagated sockeye salmon from the Redfish Lake Captive Propagation program are considered part of this species (70 FR 37160; June 28, 2005).

The NMFS has determined that this artificially propagated population is genetically no more than moderately divergent from the natural population (NMFS 2005f). Five lakes in the Stanley Basin historically contained sockeye salmon: Alturas, Pettit, Redfish, Stanley and Yellowbelly (Bjornn *et al.* 1968). It is generally believed that adults were prevented from returning to the Sawtooth Valley from 1910 to 1934 by Sunbeam Dam. Sunbeam Dam was constructed on the Salmon River approximately 20 miles downstream of Redfish Lake. Whether Sunbeam Dam was a complete barrier to adult migration remains unknown. It has been hypothesized that some passage occurred while the dam was in place, allowing the Stanley Basin population or populations to persist (Bjornn *et al.* 1968, Waples *et al.* 1991).

Adult returns to Redfish Lake during the period 1954 through 1966 ranged from 11 to 4,361 fish (Bjornn *et al.* 1968). Sockeye salmon in Alturas Lake were extirpated in the early 1900s as a result of irrigation diversions, although residual sockeye may still exist in the lake (Chapman and Witty 1993). From 1955 to 1965, the Idaho Department of Fish and Game eradicated sockeye salmon from Pettit, Stanley, and Yellowbelly lakes, and built permanent structures on each of the lake outlets that prevented re-entry of anadromous sockeye salmon (Chapman and Witty 1993). In 1985, 1986, and 1987, 11, 29, and 16 sockeye, respectively, were counted at the Redfish Lake weir (Good *et al.* 2005b). Only 18 natural origin sockeye salmon have returned to the Stanley Basin since 1987. During the fall of 1990, no fish were observed at Lower Granit Dam or entering the lake and only one fish was observed in each of the two previous years. The first adult returns from the captive broodstock program returned to the Stanley Basin in 1999. From 1999 through 2005, a total of 345 captive brood program adults that had migrated to the ocean returned to the Stanley Basin.

Recent annual abundances of natural origin sockeye salmon in the Stanley Basin have been extremely low. No natural origin anadromous adults have returned since 1998 and the abundance of residual sockeye salmon in Redfish Lake is unknown. This species is entirely supported by adults produced through the captive propagation program at the present time. Current smolt-to-adult survival of sockeye originating from the Stanley Basin lakes is rarely greater than 0.3% (Hebdon *et al.* 2004). The status of this ESU is extremely precarious, such that there was unanimous consent among the biological review team members that the species remains in danger of extinction (Good *et al.* 2005b). However, the most recent run abundances from 2008 and 2009 have been more robust, with 650 and 809 individuals, respectively, returning to the Sawtooth Hatchery (NMFS 2011l).

Critical habitat. Critical habitat for these salmon was designated on December 28, 1993 (58 FR 68543), and encompasses the waters, waterway bottoms, and adjacent riparian zones of specified lakes and river reaches in the Columbia River that are or were accessible to listed Snake River salmon (except reaches above impassable natural falls, and Dworshak and Hells Canyon Dams). Adjacent riparian zones are defined as those areas within a horizontal distance of 300 feet from the normal line of high water of a stream channel or from the shoreline of a standing body of water. Designated critical habitat includes the Columbia River from a straight line connecting the west end of the Clatsop jetty (Oregon side) and the west end of the Peacock jetty (Washington side) and including all river reaches from the estuary upstream to the confluence of the Snake River, and all Snake River reaches upstream to the confluence of the Salmon River; all Salmon River reaches to Alturas Lake Creek; Stanley, Redfish, Yellow Belly, Pettit, and Alturas Lakes (including their inlet and outlet creeks); Alturas Lake Creek and that portion of Valley Creek between Stanley Lake Creek and the Salmon River. Critical habitat also includes all river lakes and reaches presently or historically accessible to Snake River sockeye salmon. These habitats are critical for the conservation of the species because it provides spawning and juvenile rearing habitat, areas for juvenile growth and development, and migration corridors for smolts to the ocean and adults to spawning habitat from the Pacific Ocean. Limiting factors identified for Snake River sockeye include: reduced tributary stream flow, impaired tributary passage and blocks to migration, and mainstem Columbia River hydropower system mortality.

Species-specific threats and limitations to recovery. The key factor limiting recovery of Snake River sockeye salmon ESU is survival outside of the Stanley Basin. Portions of the migration corridor in the Salmon River are impeded by water quality and temperature (IDEQ 2011). Increased temperatures may reduce the survival of adult sockeye returning to the Stanley Basin. The natural hydrological regime in the upper mainstem Salmon River Basin has been altered by water withdrawals. In most years, sockeye adult returns to Lower Granite suffer catastrophic losses (*e.g.*, >50% mortality in one year; (Reed *et al.* 2003)) before reaching the Stanley Basin, although the factors causing these losses have not been identified. In the Columbia and lower Snake River migration corridor, predation rates on juvenile sockeye salmon are unknown, but terns and cormorants consume 12% of all salmon smolts reaching the estuary, and piscivorous fish consume an estimated 8% of migrating juvenile salmon (NMFS 2011l).

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 (Moyle 1976a, Stolz and Schnell 1991, NMFS 1997b, Good *et al.* 2005a, Good *et al.* 2005b). 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.* 1996d). 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 (Nickelson *et al.* 1992, Busby *et al.* 1996d). 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.* 1996d).

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 (Nickelson *et al.* 1992, Busby *et al.* 1996d). 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 (Nickelson *et al.* 1992, Busby *et al.* 1996d), 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.* 1996d). 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).

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 2011l). 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 2011l).

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.

Central California coast steelhead

Distribution. The Central California Coast steelhead DPS includes all naturally spawned anadromous steelhead populations below natural and manmade impassable barriers in California streams from the Russian River (inclusive) to Aptos Creek (inclusive), and the drainages of San Francisco, San Pablo, and Suisun Bays eastward to Chipps Island at the confluence of the Sacramento and San Joaquin Rivers (Table 18). Tributary streams to Suisun Marsh including Suisun Creek, Green Valley Creek, and an unnamed tributary to Cordelia Slough (commonly referred to as Red Top Creek), excluding the Sacramento-San Joaquin River Basin, as well as two artificial propagation programs.

Table 18. Central California Coast steelhead populations and selected measures of population viability.

Basin	Historical abundance	Most recent spawner abundance	Hatchery abundance contributions
Russian River	65,000 (1970)	1,750-7,000 (1994)	Unknown
Lagunitas	Unknown	400-500 (1990s)	Unknown
San Gregorio	1,000 (1973)	Unknown	Unknown
Waddell Creek	481	150 (1994)	Unknown
Scott Creek	Unknown	<100 (1991)	Unknown
San Vicente Creek	150 (1982)	50 (1994)	Unknown
San Lorenzo River	20,000	<150 (1994)	Unknown
Soquel Creek	500-800 (1982)	<100 (1991)	Unknown
Aptos Creek	200 (1982)	50-75 (1994)	Unknown
Total	**94,000**	**2,400-8,125**	

The DPS is entirely composed of winter run fish. Adults migrate upstream from December-April, and smolts emigrate between March-May (Shapovalov and Taft 1954, Hayes *et al.* 2008). At the time of the 1996 status review and 1997 listing, little information was available on the specific demographics and life history characteristics of steelhead in this DPS. While age at smoltification typically ranges from 1 to 4 years, recent studies by Sogard *et al.* (2009) that growth rates in Soquel Creek likely prevent juveniles from undergoing smoltification until age 2. Survival in freshwater reaches tends to be higher in summer and lower from winter through spring for year classes 0 and 1 (Sogard *et al.* 2009). Larger individuals also survive more readily than do smaller fish within year classes (Sogard *et al.* 2009). Greater movement of juveniles in freshwater has been observed in winter and spring versus summer and fall time periods, with smaller individuals more likely to move between stream areas (Sogard *et al.* 2009). Growth rates during this time have rarely been observed to exceed 0.3 mm per day and are highest in winter through spring, potentially due to higher water flow rates and greater food availability (Boughton *et al.* 2007, Hayes *et al.* 2008, Sogard *et al.* 2009).

Status and trends. The Central California Coast steelhead DPS was listed as a threatened species on August 18, 1997 (62 FR 43937) and was reaffirmed on January 5, 2006 (71 FR 834). Table 19 identifies runs within the Central California Coast steelhead DPS and their estimated run sizes.

Table 19. Central California coast steelhead populations and their estimated abundances.

Basin	Estimated abundance[a]	Year
Russian River	65,000	1970
	1,750-7,000	1994
Lagunitas	500	1994
	400-500	1990s
San Gregorio	1,000	1973
Waddell Creek	481	1933-1942
	250*	1982
	150*	1994
Scott Creek	400	1991
	<100	1991
	300	1994
San Vicente Creek	150*	1982
	50*	1994
San Lorenzo River	20,000	Pre 1965
	1,614	1977
	>3,000*	1978
	600	1979
	3,000	1982
	“few”	1991
	<150*	1994
Soquel Creek	500-800*	1982
	<100	1991
	50-100*	1994
Aptos Creek	200*	1982
	<100	1991
	50-75*	1994

[a] A complete list of data sources is available in Good *et al.* 2005. According to Good *et al.* the basis for certain estimates is questionable (noted with an asterisk above).

Estimates of historical abundance are provided here only for background, as the accuracy of the estimates is unclear. An estimate of historical abundance for the total DPS is provided by CDFG at 94,000 fish. This estimate is based on a partial data set and “best professional judgment” (see

(Good *et al.* 2005b) for a discussion). Other estimates of historical abundance are on a per river basis. According to Busby *et al.* (1996a), Shapovalov and Taft (1954) described an average of about 500 adults in Waddell Creek (Santa Cruz County) for the 1930s and early 1940s, whereas Johnson (1964) estimated a run size of 20,000 steelhead in the San Lorenzo River before 1965. Most of the estimates for run sizes within the DPS are more recent (Table 26). Two rivers thought to have contained the largest populations within the DPS were the Russian River, and the San Lorenzo River. Based on run size estimates from the 1990s, the Russian River is still likely the largest run within the DPS, albeit estimates suggest the population has declined between 90-96% from the 1970 level.

No current estimates of total population size are available for this DPS, and consequently there is no time series data available to evaluate the central California coast steelhead population trends. Rather, a general dearth of data on adult steelhead within the DPS led the biological review team to examine data collected on juvenile steelhead (Good *et al.* 2005b). In general, juvenile data is considered a poor indicator of the reproductive portion of the population as juvenile age classes exhibit greater mortality rates, which are closely tied to stochastic events, and may move widely within a basin (which may include intermixing with other populations). There is no simple relationship between juvenile and adult numbers (Shea and Mangel 2001). Nonetheless, there was not enough adult data upon which the biological review team could base an assessment of the population trends within the DPS. Therefore, the biological review team log-transformed and normalized juvenile survey data from a number of watersheds (presumed populations). As a result, the team derived trend estimates for five populations: the San Lorenzo River, Scott Creek, Waddell Creek, Gazos Creek, and Redwood Creek in Marin County (see (Good *et al.* 2005b) for a detailed discussion of the approach). All populations exhibited downward trends in abundance. Accordingly, provided the juvenile data is representative of the true trend, this data suggests that there is an overall downward trend in abundance in the DPS.

In the most recent review of the status of this DPS, most members of the biological review team (69%) considered this DPS likely to become endangered thus supporting the renewal of the threatened status for central California coast steelhead. Notably, 25% of the team voted that the DPS be upgraded to endangered status (voted the DPS as in danger of extinction)(Good *et al.* 2005b). Abundance and productivity were of relatively high concern (as a contributing factor to risk of extinction), and spatial structure was also of concern.

Since the original status review, fishing regulations have changed that probably reduces extinction risk for Central California Coast steelhead. Ocean sport harvest is prohibited, and ocean harvest is considered rare. Although freshwater streams are closed to fishing year round, CDFG has identified certain streams as exceptions where they allow catch-and-release angling or summer trout fishing. In catch-and-release streams, all wild steelhead must be released unharmed.

Critical habitat. Critical habitat was designated for the Central California Coast steelhead DPS on September 2, 2005 (70 FR 52488), and includes areas within the following hydrologic units: Russian River, Bodega, Marin Coastal, San Mateo, Bay Bridge, Santa Clara, San Pablo, and Big Basin. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding habitat. The critical habitat designation for this ESU identifies primary constituent elements that include sites necessary to support one or more steelhead life stages. Specific sites 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. The critical habitat designation (70 FR 52488) contains additional details on the sub-areas that are included as part of this designation, and the areas that were excluded from designation.

In total, Central California Coast steelhead occupy 46 watersheds (freshwater and estuarine). The total area of habitat designated as critical includes about 1,500 miles of stream habitat and about 400 square miles of estuarine habitat (principally Humboldt Bay). 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. In estuarine areas the lateral extent is defined by the extreme high water because extreme high tide areas encompass those areas typically inundated by water and regularly occupied by juvenile salmon during the spring and summer, when they are migrating in the nearshore zone and relying on cover and refuge qualities provided by these habitats, and while they are foraging. Of the 46 occupied watersheds reviewed in NMFS' assessment of critical habitat, 14 watersheds received a low rating of conservation value, 13 received a medium rating, and 19 received a high rating of conservation value for the species.

California Central Valley steelhead

Distribution. California Central Valley steelhead occupy the Sacramento and San Joaquin Rivers and their tributaries, although they were once widespread throughout the Central Valley (Busby *et al.* 1996a, Zimmerman *et al.* 2008). Steelhead were found from the upper Sacramento and Pit River systems (now inaccessible due to Shasta and Keswick Dams), south to the Kings and possibly the Kern River systems (now inaccessible due to extensive alteration from water diversion projects), and in both east- and west-side Sacramento River tributaries (Yoshiyama *et al.* 1996). The present distribution has been greatly reduced (McEwan and Jackson 1996a). The CACSS (1988) reported a reduction of steelhead habitat from 6,000 miles historically to 300 miles today. Historically, steelhead probably ascended Clear Creek past the French Gulch area, but access to the upper basin was blocked by Whiskeytown Dam in 1964 (Yoshiyama *et al.* 1996). Steelhead also occurred in the upper drainages of the Feather, American, Yuba, and Stanislaus Rivers which are now inaccessible (McEwan and Jackson 1996a, Yoshiyama *et al.* 1996).

Existing wild steelhead populations in the Central Valley are mostly confined to the upper Sacramento River and its tributaries, including Antelope, Deer, and Mill Creeks and the Yuba River. Populations may exist in Big Chico and Butte Creeks and a few wild steelhead are produced in the American and Feather Rivers (McEwan and Jackson 1996a). Recent snorkel surveys (1999 to 2002) indicate that steelhead are present in Clear Creek (Good *et al.* 2005a). Because of the large resident *O. mykiss* population in Clear Creek, steelhead spawner abundance has not been estimated. Until recently, steelhead were thought to be extirpated from the San Joaquin River system. Recent monitoring has detected small self-sustaining populations of steelhead in the Stanislaus, Mokelumne, Calaveras, and other streams previously thought to be devoid of steelhead (McEwan 2001). On the Stanislaus River, steelhead smolts have been captured in rotary screw traps at Caswell State Park and Oakdale each year since 1995 (Demko and Cramer 2000). It is possible that naturally spawning populations exist in many other streams but are undetected due to lack of monitoring programs (IEPSPWT 1999). Coleman and Feather River hatcheries were included in the DPS in 2006 (NMFS 2011c).

The Sacramento and San Joaquin Rivers offer the only migration route to the drainages of the Sierra Nevada and southern Cascade mountain ranges for anadromous fish. The CDFG considers all steelhead in the Central Valley as winter steelhead, although “three distinct runs,” including summer steelhead, may have occurred there as recently as 1947 (McEwan and Jackson 1996a)(CDFG 1995). Steelhead in these basins travel extensive distances in freshwater (some exceed 300 km to their natal streams), making these the longest freshwater migrations of any population of winter steelhead. The upper Sacramento River essentially receives a single continuous run of steelhead from July through May, with peaks in September and February. Spawning begins in late December and can extend into April (McEwan and Jackson 1996b).

Status and trends. The NMFS originally listed California Central Valley steelhead as threatened in 1998; this status was reviewed and retained on January 5, 2006 (71 FR 834). Historic run size is difficult to estimate given the paucity of data, but may have approached one to two million adults annually (McEwan 2001). By the early 1960s, the steelhead run size had declined to about 40,000 adults (McEwan 2001). Over the past 30 years, the naturally spawned steelhead populations in the upper Sacramento River have declined substantially. Hallock *et al.* (1961) estimated an average of 20,540 adult steelhead occurred in the Sacramento River (upstream of the Feather River). Steelhead counts at Red Bluff Diversion Dam declined from an average of 11,187 for the period of 1967 to 1977, to an average of approximately 2,000 through the early 1990s, with an estimated total annual run size for the entire Sacramento-San Joaquin system at no more than 10,000 adults (based on Red Bluff Diversion Dam counts; (McEwan and Jackson 1996a, McEwan 2001). The five-year geometric mean, however, is just under 2,000 steelhead (Table 20), and the long-term trend suggests that the population is declining.

Table 20. California Central Valley steelhead and their long-term trend**.**

Population	**5-year mean (min – max)**[a]	**λ**	**Long-term trend**[b]
Sacramento River	1,952 (1,425-12,320)	0.95 (0.90,1.02)	-0.09 (-0.13,-0.06)

[a]Refers to the period ending in 1993, when steelhead counts at Red Bluff Diversion dam ended. Data reported in Good *et al.* 2005.
[b] 90% confidence limits in parentheses.

The only consistent data available on steelhead numbers in the San Joaquin River basin come from CDFG mid-water trawling samples collected on the lower San Joaquin River at Mossdale. These data indicate a decline in steelhead numbers in the early 1990s, which have remained low through 2002 (Good *et al.* 2005a). In 2004, a total of 12 steelhead smolts were collected at Mossdale (Good *et al.* 2005a).

Based upon information since 2005, Central Valley steelhead populations have generally declined, with hatchery fish forming a larger portion of populations (NMFS 2011c). Battle Creek surveys have documented relatively stable counts since 2005 after a precipitous drop for three years prior to that. Returns to Coleman hatchery have varied widely from year to year, with no clear trend. An average of 151 redds have been found in Clear Creek from 2001 to 2010, with similar numbners (154) in American River over nearly the same time frame. A very small number (40-70) of redds have been counted over the past three years in the Lower Mokelumne River, but the trend is on the rise. Returns to the Feather River Hatchery have plummeted from 679 to 86 fish from 2008 to 2010.

Reynolds *et al.* (1993) reported that 95% of salmonid habitat in California's Central Valley has been lost, largely due to mining and water development activities. They also noted that declines are "due mostly to water development, inadequate instream flows, rapid flow fluctuations, high summer water temperatures in streams immediately below reservoirs, diversion dams which block access, and entrainment of juveniles into unscreened or poorly screened diversions." Thus, overall habitat problems in this ESU relate primarily to water development resulting in inadequate flows, flow fluctuations, blockages, and entrainment into diversions (McEwan and Jackson 1996a). Other problems related to land use practices (agriculture and forestry) and urbanization have also contributed to population declines. It is unclear how harvest has affected California's Central Valley steelhead, although it is likely a continuing threat. A CDFG creel census in 2000 indicated that most fish are caught and released, but due to the size of the catch and release fishery (more than 14,000 steelhead were caught and released according to the survey) even a small amount of mortality in this fishery could cause declines in the populations.

Critical habitat. The NMFS designated critical habitat for California Central Valley steelhead on September 2, 2005 (70 FR 52488). Specific geographic areas designated include the following CALWATER hydrological units: Tehama, Whitmore, Redding, Eastern Tehama, Sacramento Delta, Valley-Putach-Cache, American River, Marysville, Yuba, Valley American, Colusa Basin, Butte Creek, Ball Mountain, Shata Bally, North Valley Floor, Upper Calaveras, Stanislaus River, San Joaquin Valley, Delta-Mendota Canal, North Diablo Range, and the San Joaquin Delta. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding habitat. The critical habitat designation for this ESU identifies primary constituent elements that include sites necessary to support one or more steelhead life stages. Specific sites 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. The critical habitat designation (70 FR 52488) contains additional details on the sub-areas that are included as part of this designation, and the areas that were excluded from designation.

In total, California Central Valley steelhead occupy 67 watersheds (freshwater and estuarine). The total area of habitat designated as critical includes about 2,300 miles of stream habitat and about 250 square miles of estuarine habitat in the San Francisco-San Pablo-Suisan Bay estuarine complex. 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. In estuarine areas the lateral extent is defined by the extreme high water because extreme high tide areas encompass those areas typically inundated by water and regularly occupied by juvenile salmon during the spring and summer, when they are migrating in the nearshore zone and relying on cover and refuge qualities provided by these habitats, and while they are foraging. Of the 67 watersheds reviewed in the NMFS' assessment of critical habitat, seven watersheds received a low rating of conservation value, three received a medium rating, and 27 received a high rating of conservation value for the species.

Lower Columbia River steelhead

Distribution. Lower Columbia River steelhead include naturally produced steelhead returning to Columbia River tributaries on the Washington side between the Cowlitz and Wind rivers in Washington and on the Oregon side between the Willamette and Hood rivers, inclusive (Figure 17). In the Willamette River, the upstream boundary of this species is at Willamette Falls. This species includes both winter and summer steelhead. Two hatchery populations are included in this species. Table 21 identifies the populations that comprise Lower Columbia River steelhead (23 historically) and summarizes several measures available to characterize population viability.

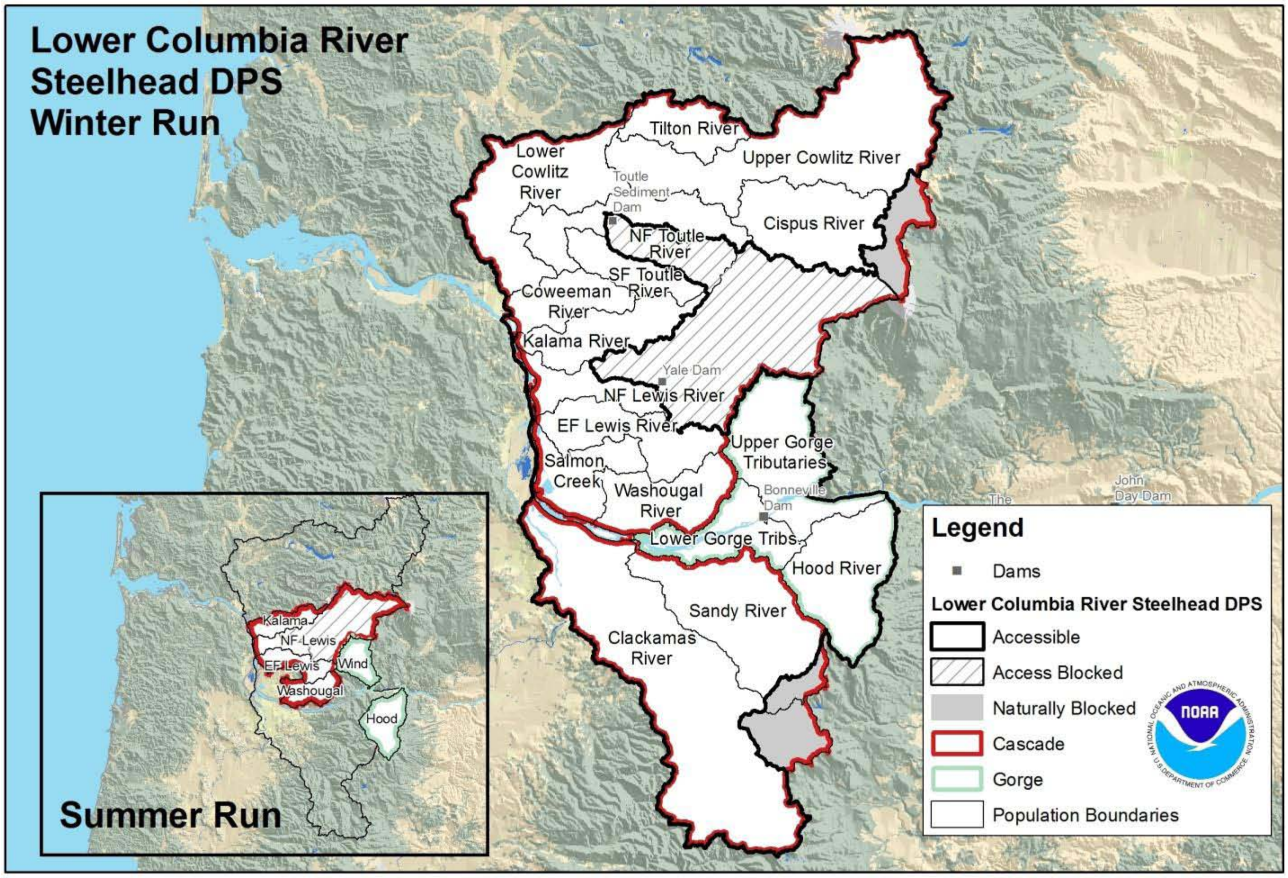

Figure 17. Population boundaries, dams, accessible areas, and extirpated reaches of lower Columbia River steelhead distribution, incorporating critical habitat boundaries (dark black line) and incorporating summer versus winter runs. Taken from NMFS (2011).

Table 21. Lower Columbia River steelhead populations and select measures of population viability.

Life history	Population	Historical abundance[a]	Mean number of spawners	Percent hatchery contribution	Median short-term growth rate (λ)[b]
Winter	Cispus River		2,787[c]	73	
	Tilton River				
	Upper Cowlitz River				
	Lower Cowlitz River	1,672			

Life history	Population	Historical abundance[a]	Mean number of spawners	Percent hatchery contribution	Median short-term growth rate (λ)[b]
	Coweeman River	2,243	466[d]	50	0.920, 0.787
	South Fork Toutle River	2,627	504[d]	2	0.933, 0.929
	North Fork Toutle River	3,770	196[d]	0	1.038, 1.038
	Kalama River	554	726[d]	0	0.984, 0.922
	North Fork Lewis River	713			
	East Fork Lewis River	3,131			
	Salmon Creek				
	Washougal River	2,497	323[d]	0	
	Clackamas River		560[e]	41	0.875, 0.830
	Sandy River		977e	42	0.866, 0.782
	Lower Columbia Gorge tributaries	793			
	Upper Columbia Gorge tributaries	243			
	Hood River		756[f]	52	
Summer	Wind River	2,288	472[g]	5	0.995, 0.903
	Hood River		931[f]	83	Unknown
	Washougal River	1,419	264[g]	8	1.029, 0.960
	East Fork Lewis River	422	434[g]	25	
	North Fork Lewis River				
	Kalama River	3,165	474[g]	32	0.900, 0.664

[a] All data reported by Good *et al.* 2005. Estimate of historical abundance derived through EDT model associated with large uncertainty. Model also incorporates presently available habitat that was not historically available and vice versa.

[b] λ calculation assumed either hatchery fish fail to reproduce or reproduce at the rate of wild individuals, respectively.

[c] Data from 2002.

[d] Data from 1998-2002.

[e] Data from 1997-2001.

[f] Data from 1996-2000.

[g] Data from 1999-2003.

Summer steelhead return sexually immature to the Columbia River from May to November, and spend several months in freshwater prior to spawning. Winter steelhead enter freshwater from November to April, are close to sexual maturation during freshwater entry, and spawn shortly after arrival in their natal streams. Where both races spawn in the same stream, summer steelhead tend to spawn at higher elevations than the winter forms.

Status and trends. The NMFS listed Lower Columbia River steelhead as threatened on March 19, 1998 (63 FR 13347), and reaffirmed their status as threatened on January 5, 2006 (71 FR 834). The 1998 status review noted that this ESU is characterized by populations at low abundance relative to historical levels, significant population declines since the mid-1980s, and widespread occurrence of hatchery fish in naturally spawning steelhead populations. During this review the NMFS was unable to identify any natural populations that would be considered at low risk.

All populations declined between 1980 and 2000, with sharp declines beginning in 1995. Those with adequate data for modeling are estimated to have a high extinction risk (Good *et al.* 2005b). Abundance trends are generally negative, showing that most populations are in decline, although some populations, particularly summer run, have shown higher return in the last 2 to 3 years. Historical counts in some of the larger tributaries (Cowlitz, Kalama, and Sandy Rivers) suggest the population probably exceeds 20,000 fish while in the 1990s fish abundance dropped to 1,000 to 2,000. Recent abundance estimates of natural-origin spawners range from completely extirpated for some populations above impassable barriers to over 700 for the Kalama and Sandy winter-run populations. A number of the populations have a substantial fraction of hatchery-origin spawners in spawning areas, and are hypothesized to be sustained largely by hatchery production. Exceptions are the Kalama, the Toutle, and East Fork Lewis winter-run populations. These populations have relatively low recent mean abundance estimates with the largest being the Kalama (geometric mean of 728 spawners). Since 2000, abundance for most populations increased, peaking in 2004 before returning to lower levels near the historical mean (NMFS 2011d). No sustained improvement in abundance has been observed since this time (NMFS 2011d). However, hatchery releases have greatly increased from two million to three million fish annually (NMFS 2011d). Currently, 16 of 23 populations are considered at high risk of extinction.

Critical habitat. The NMFS designated critical habitat for Lower Columbia River steelhead on September 2, 2005 (70 FR 52630). Designated critical habitat includes the following subbasins: Middle Columbia/Hood subbasin, Lower Columbia/Sandy subbasin, Lewis subbasin, Lower Columbia/Clatskanie subbasin, Upper Cowlitz subbasin, Cowlitz subbasin, Clackamas subbasin, Lower Willamette subbasin, and the Lower Columbia River corridor. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding habitat. The critical habitat designation for this DPS identifies primary constituent elements that include sites necessary to support one or more steelhead life stages. Specific sites 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. The critical habitat designation (70 FR 52630) contains additional description of the watersheds that are included as part of this designation, and any areas specifically excluded from the designation.

In total, Lower Columbia River steelhead occupy 32 watersheds. The total area of habitat designated as critical includes about 2,340 miles of stream habitat. This designation includes the stream channels within the designated stream reaches, and 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. Of the 32 watersheds reviewed in NMFS' assessment of critical habitat, two watersheds received a low rating of conservation value, 11 received a medium rating, and 26 received a high rating of conservation value for the species. Limiting factors identified for Lower Columbia River steelhead include: degraded floodplain and steam channel structure and function, reduced access to spawning/rearing habitat, altered stream flow in tributaries, excessive sediment and elevated water temperatures in tributaries, and hatchery impacts.

Species-specific threats and limitations to recovery. Several direct and habitat-related factors inhibit the recovery of Lower Columbia River Coho salmon (LCFRB 2010, NMFS 2011d). Habitat impacts include degraded estuarine and near-shore marine habitat resulting from cumulative impacts of land use and flow management by the Columbia River hydropower system, fish passage barriers that limit access to spawning and rearing habitats, floodplain connectivity and function, channel structure and complexity, riparian areas and large wood supply, stream substrate, stream flow, and water quality have been degraded as a result of cumulative impacts of agriculture, forestry, and development. Additional threats to habitat include an altered flow regime and Columbia River plume has altered the temperature regime and estuarine food web, and has reduced ocean productivity, reduced access to off-channel rearing habitat in the lower Columbia River, as well as reduced productivity resulting from sediment and nutrient-related changes in the estuary. Other impacts also include hatchery- and harvest-related effects, juvenile fish strandings that result from ship wakes, avian and marine mammal predation in the lower mainstem Columbia River and estuary, and contaminants affecting fish health and reproduction.

Middle Columbia River steelhead

Distribution. The Middle Columbia River steelhead DPS includes all naturally spawned anadromous steelhead populations below natural and manmade impassible barriers in Oregon and Washington drainages upstream of the Hood and Wind River systems, up to and including the Yakima River (61 FR 41541)(Figure 18). Steelhead from the Snake River Basin are excluded from this DPS. Seven artificial propagation program are part of this DPS. These artificially propagated populations are considered no more divergent relative to the local natural populations than would be expected between closely related natural populations within the DPS.

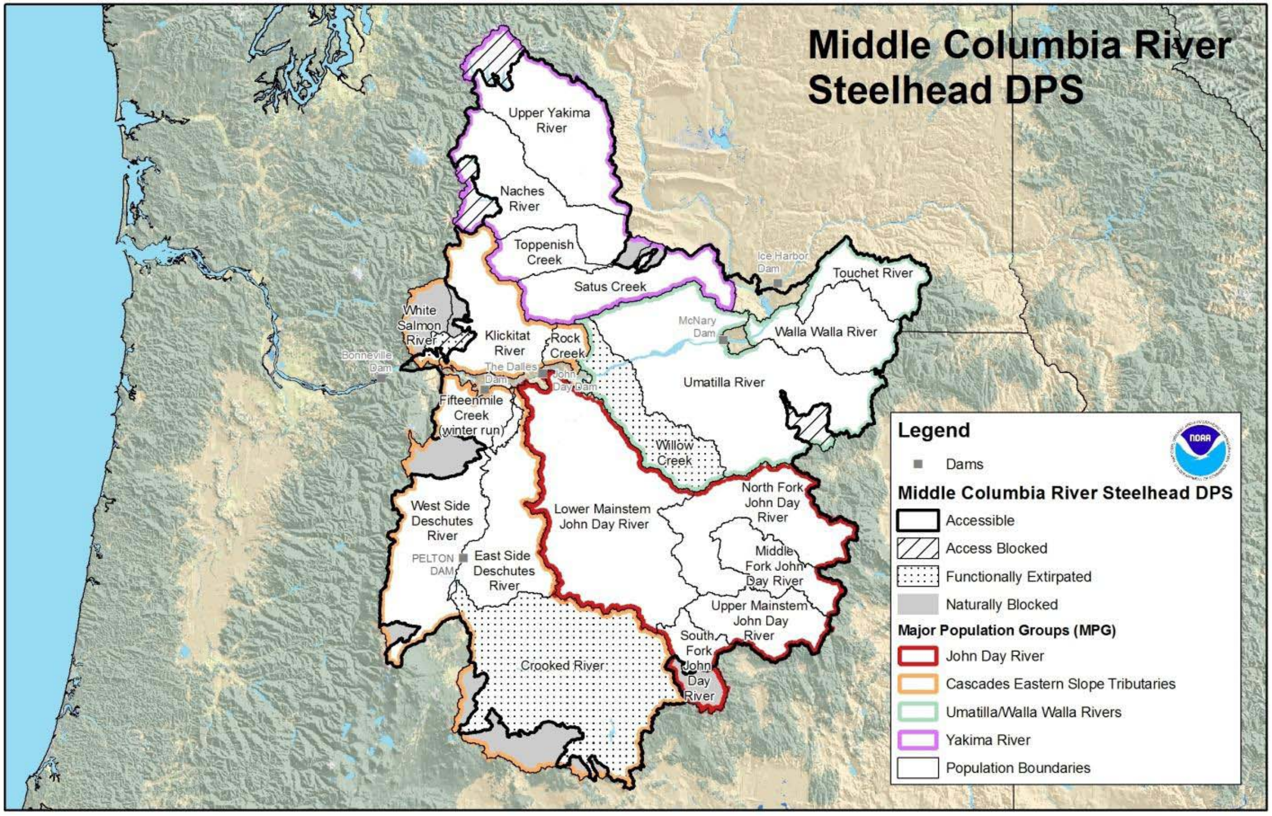

Figure 18. Population boundaries, dams, accessible areas, and extirpated reaches of lower Columbia River steelhead distribution, incorporating critical habitat boundaries (dark black line). Taken from NMFS (2011e).

Middle Columbia River steelhead occupy the intermontane region of the Pacific northwest, which includes some of the driest areas in the region generally receiving less than 40 cm of rainfall annually. Major drainages in this ESU are the Deschutes, John Day, Umatilla, Walla Walla, Yakima, and Klickitat river systems. The area is generally characterized by its dry climate and harsh temperature extremes. Almost all steelhead populations within this DPS are summer-run fish; the only exceptions are the only populations of inland winter steelhead, which occur in the Klickitat River and Fifteenmile Creek (Busby *et al.* 1996d). According to Interior Columbia Basin Technical Recovery Team (ICTRT 2003a) this DPS is comprised of 16 putative populations in four major population groups (Cascades Eastern Slopes Tributaries, John Day River, Walla Walla and Umatilla Rivers, and Yakima River) and one unaffiliated independent population (Rock Creek). Table 22 lists extant (putative) populations that compose this DPS. There are two extinct populations in the Cascades Eastern Slope major population group, the White Salmon River and Deschutes Crooked River above the Pelton/Round Butte Dam complex. Present population structure is delineated largely on the basis of geographical proximity, topography, distance, ecological similarities or differences. Additional genetic studies are needed to describe the DPS substructure, as well as the fine-scale genetic structure of the populations within a particular basin (e.g., John Day River).

Table 22. Middle Columbia River steelhead populations and select measures of population viability.

Population[a]	**Major population groups**	**Mean number of spawners (range)**[b]	**Percent hatchery contribution**[c]	**Long-term growth rate (λ)**[d]
Klickitat River	Cascade Eastern Slope	155 redds (97-261)		
Fifteenmile Creek	Cascade Eastern Slope	2.87 rpm (1.3-6.0)	0	1.129
Deschutes River - eastside	Cascade Eastern Slope	13,455 (10,026-21,457)	72	1.022,0.840, 0.942
Descutes River – westside	Cascade Eastern Slope			
John Day lower mainstem tributaries	John Day River	1.4 rpm (0-5.4)		1.013
North Fork John Day	John Day River	Upper NF - 2.57 rpm (1.6-5.0)[e] Lower NF - 3.52 rpm (1.5-8.8)		1.011 1.174
Middle Fork John Day	John Day River	3.70 rpm (1.7-6.2)		0.966
South Fork John Day	John Day River	2.52 rpm (0.9-8.2)		0.967
John Day upper mainstem	John Day River	2,122 (926-4,168)	4	0.975, 0.966
Rock Creek	Unaffiliated Area			
Umatilla River	Walla Walla & Umatilla	2,486 (1,480-5,157)	40	1.007, 0.969
Walla Walla	Walla Walla & Umatilla			
Touchet River	Walla Walla & Umatilla	345 (273-527)[f]	16	0.961, 0.939
Toppenish & Satus Creek	Yakima River			
Naches River	Yakima River			
Yakima River upper mainstem	Yakima River	1,801 (1,058-4,061)	3	1.009

[a] Population groups defined by the ICBTRT (2003).
[b] Values represent the 5-year geometric mean in spawners, redds, or redds per mile (RPM). Values calculated from data series using years 1997-2001 or 1998-2001. See Good *et al.* (2005) for details.
[c] Hatchery production in the recent past and at present consists of locally-derived broodstock, although straying of production fish into the Deschutes River has been a persistent problem. Data from Good *et al.* 2005.

[d]Multiple estimates for long-term growth (λ) presented for some populations representing two different assumptions on the contribution of hatchery fish to the natural production. Where two or more values are presented, the first value reflects the assumption that hatchery fish do not contribute to natural production, and the second value reflects the assumption that hatchery contribute to natural production at the same rate as natural-origin spawners. Deschutes River values are reflective of total population, not eastside only. The λ value is calculated from data (1980-1999) from Warm Springs area. Data series upon which values are calculated varies across basins. See Good *et al.* (2005) for details on the length and time of data series available by population.

Most Middle Columbia River steelhead smolt at 2 years of age and spend 1 to 2 years at sea prior to re-entering natal river systems. They may remain in such rivers for up to a year prior to spawning (Howell *et al.* 1985b). Within this ESU, the Klickitat River is unusual, as it produces both summer and winter steelhead. The summer steelhead are dominated by year-class-two ocean steelhead, whereas most other rivers in this region produce about equal numbers of both age-one and age-two ocean steelhead. Factors contributing to decline include hydropower development and agriculture; these land uses impede or prevent migrations, alter water availability, and alter water chemistry and temperatures.

Status and trends. Middle Columbia River steelhead were listed as threatened in 1999 (64 FR 14517), and their status was reaffirmed on January 5, 2006 (71 FR 834). The precise pre-1960 abundance of this species is unknown. Based upon the Washington Department of Fish and Wildlife's estimates of the historic run size for the Yakima River at 100,000 steelhead, Busby *et al.* (1996d) surmised that total DPS abundance likely exceeded 300,000 returning adults. By 1993, the estimated 5-year average size (ending in 1993) of the Middle Columbia steelhead DPS was 142,000 fish (Busby *et al.* 1996d). Survey data collected between 1997 and 2001 indicates that several populations within the DPS have increased since the last status review (Good *et al.* 2005b). However, the long-term annual population growth rate (λ) is less than one for most populations (see Table 29).

In contrast, short term trends in major areas were positive for 7 of the 12 areas with available data (Good *et al.* 2005b). Spawner numbers in the Yakima River, the Deschutes River and sections of the John Day River system were substantially higher compared to numbers surveyed between 1992 and 1997 (Good *et al.* 2005b). Similarly, spawner numbers substantially increased in the Umatilla River and Fifteenmile Creek relative to annual levels in the early 1990s. Nonetheless, most populations remain below interim target levels. For instance, the Yakima River returns are still substantially below interim target levels of 8,900 (the current 5-year average is 1,747 fish) and estimated historical return levels. In fact, the majority of spawning occurs in only one tributary, Satus Creek (Berg 2001). Based on recent 5-year geometric means, only the Deschutes River exceeded interim target levels (Good *et al.* 2005b). While increases in short-term trends could suggest improvements within the DPS, given that the average population growth rate across all streams is negative (0.98 assuming hatchery spawners do not contribute to production, and 0.97 assuming that both hatchery and natural-origin fish contribute equally) and evidence of large fluctuation in marine survival for the species, recent increases in population sizes must be viewed cautiously.

Recent data indicate varied trends among the population groups (NMFS 2011e). Monitored populations with the Cassades Eastern Slope Tributaries group show declining trends since 2005, although natural origin spawners are composing a higher proportion of total retuns. On the contrary, John Day River populations seem to be improving. Populations in the Yakima River group also improved, although abundance remains very low. Two populations within the Umatilla/Walla Walla Rivers group improved, while one remained stable.

Critical habitat. The NMFS designated critical habitat for Middle Columbia River steelhead on September 2, 2005 (70 FR 52630). Designated critical habitat includes the following subbasins: Upper Yakima, Naches, Lower Yakima, Middle Columbia/Lake Wallula, Walla Walla, Umatilla, Middle Columbia/Hood, Klickitat, Upper John Day, North Fork John Day, Middle Fork John Day, Lower John Day, Lower Deschutes, Trout, and the Upper Columbia/Priest Rapids subbasins, and the Columbia River corridor. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding habitat. The critical habitat designation for this DPS identifies primary constituent elements that include sites necessary to support one or more steelhead life stages. Specific sites 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. The final rule (70 FR 52630) lists the watersheds that comprise the designated subbasins and any areas that are specifically excluded from the designation.

In total, there are 114 watersheds within the range of Middle Columbia River steelhead. The total area of habitat designated as critical includes about 5,800 miles of stream habitat. 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. Of the 114 watersheds reviewed in NMFS' assessment of critical habitat for Middle Columbia River steelhead, nine watersheds received a low rating of conservation value, 24 received a medium rating, and 81 received a high rating of conservation value for the species. Although pristine habitat conditions are still present in some wilderness, roadless, and undeveloped areas, habitat complexity has been greatly reduced in many areas of designated critical habitat for Middle Columbia River steelhead. Limiting factors identified for Middle Columbia River steelhead include: hydropower system mortality, reduced stream flow, impaired passage, excessive sediment, degraded water quality, and altered channel morphology and floodplain.

Species-specific threats and limitations to recovery. Floodplain connectivity and function, channel structure and complexity, riparian areas, fish passage, stream substrate, stream flow, and water quality have been degraded as a result of cumulative impacts of agriculture, forestry, tributary hydro system activities, and development impair recovery of Middle Columbia River steelhead (NMFS 2009b, 2011e). Additional problems for recovery are posed by hatchery- and harvest-related effects, predation, competition, disease, degraded estuarine and nearshore habitats, and the influence of mainstem Columbia River hydropower facilities.

Upper Columbia River steelhead

Distribution. Upper Columbia River steelhead occupy the Columbia River Basin upstream from the Yakima River, Washington to the border between the US and Canada (Figure 19). This area includes the Wenatchee, Entiat, and Okanogan Rivers (Table 23). All Upper Columbia River steelhead are summer steelhead. Steelhead primarily use streams of this region that drain the northern Cascade Mountains of Washington State. This species includes hatchery populations of summer steelhead from the Wells Hatchery because it probably retains the genetic resources of steelhead populations that once occurred above the Grand Coulee Dam. This species does not include the Skamania Hatchery stock because of its non-native genetic heritage.

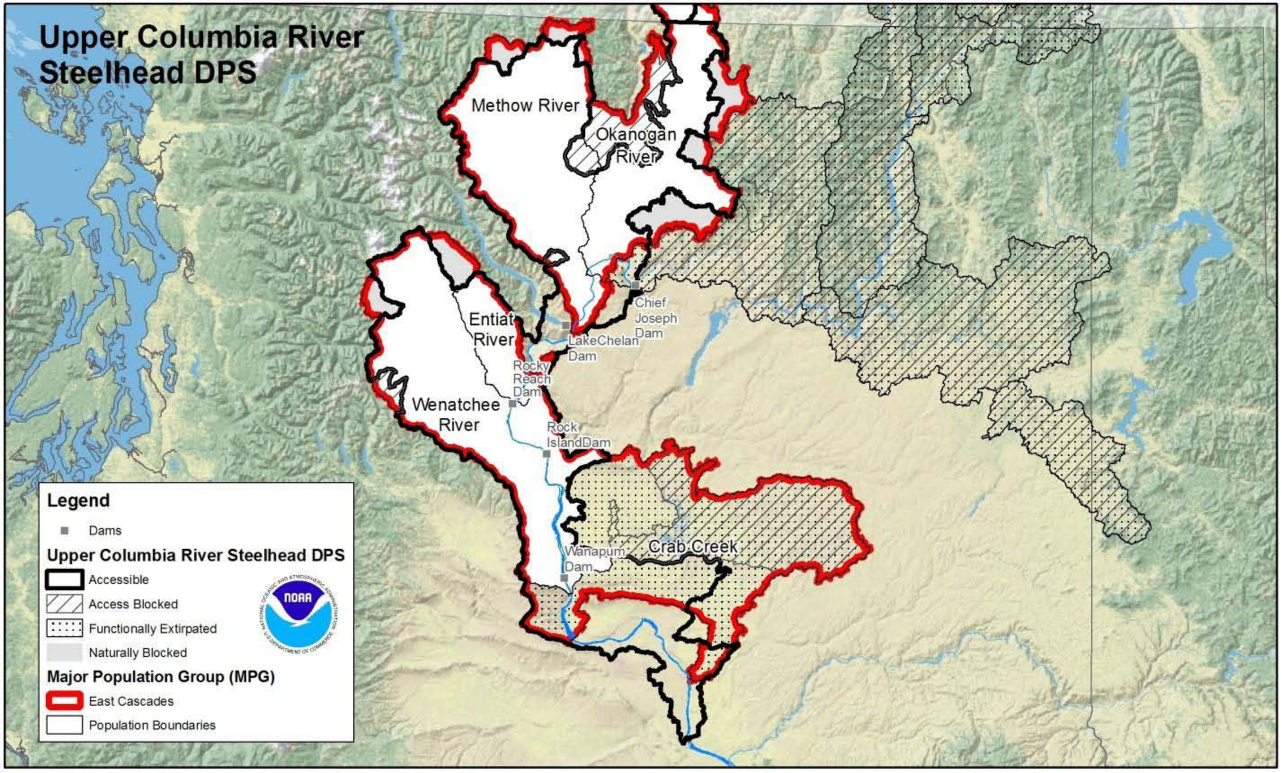

Figure 19. Population boundaries, dams, accessible areas, and extirpated reaches of upper Columbia River steelhead distribution, incorporating critical habitat boundaries (dark black line). Taken from NMFS (NMFS 2011i).

Columbia River steelhead life histories are complex. Adults return in late summer and early fall, with most migrating relatively quickly to their natal tributaries. A portion of the returning run overwinters in mainstem reservoirs, passing over upper- to mid-Columbia dams in April and May of the following year. Spawning occurs in the late spring of the year following river entry. Juvenile steelhead spend one to seven years rearing in freshwater before migrating to sea. Smolt outmigrations are predominantly year class two and three (juveniles). Most adult steelhead return after from sea after one or two years.

Table 23. Upper Columbia River steelhead salmon populations, abundances, and hatchery contributions (Good *et al.* 2005b).

Population	Historical abundance	Most recent spawner abundance	Hatchery abundance contributions
Wenatchee/Entiat rivers	Unknown	1,899-8,036	71%
Methow/Okanogan rivers	Unknown	1,879-12,801	91%
Total	**Unknown**	**3,778-20,837**	

Status and trends. Upper Columbia River steelhead were originally listed as endangered in 1997 (62 FR 43937), after their status was reviewed, they were reclassified to threatened on January 5, 2006 and then reinstated to endangered status per US District Court decision in June 2007 (62 FR 43937).

Abundance estimates of returning naturally produced Upper Columbia River steelhead have been based on extrapolations from mainstem dam counts and associated sampling information (e.g., hatchery/wild fraction, age composition). Returns of both hatchery and naturally produced steelhead to the upper Columbia River have increased in recent years. The average 1997-2001 return through the Priest Rapids fish ladder was approximately 12,900 fish. The average for the previous five years (1992 to 1996) was 7,800 fish. Abundance estimates of returning naturally produced Upper Columbia River steelhead have been based on extrapolations from mainstem dam counts and associated sampling information (e.g., hatchery/wild fraction, age composition). The natural component of the annual steelhead run over Priest Rapids Dam increased from an average of 1,040 (1992-1996), representing about 10% of the total adult count, to 2,200 (1997-2001), representing about 17% of the adult count during this period of time (ICBTRT 2003). Since 2005, abundance estimates have increased in all four populations within the ESU, although a large portion of returns are hatchery origin fish (NMFS 2011i).

In terms of natural production, recent population abundances for the Wenatchee, Entiat, and the Methow population remain well below the minimum abundance thresholds developed for these populations (ICBTRT 2005). A five-year geometric mean (1997 to 2001) of approximately 900 naturally produced steelhead returned to the Wenatchee and Entiat rivers (combined). Although this is well below the minimum abundance thresholds, it represents an improvement (an increasing trend of 3.4% per year). However, the average percentage of natural fish for the recent five-year period dropped from 35% to 29%, compared to the previous status review. For the Methow population, the five-year geometric mean of natural returns over Wells Dam was 358. Although this is well below the minimum abundance threshold, it is increasing trend of 5.9% per year. In addition, the 2001 return (1,380 naturally produced spawners) was the highest single annual return in the 25-year data series. However, the average percentage of wild origin spawners dropped from 19% for the period prior to the 1998 status review to 9% for the 1997 to 2001 returns. This DPS is failing to meet viability criteria in all four categories; productivity, abundance, spatial structure, and genetic diversity. Overall, long-term population growth is in decline. Returns are predominantly hatchery origin and all Upper Columbia River steelhead populations have reduced genetic diversity from homogenization of populations that occurred during the Grand Coulee Fish Maintenance project from 1939-1943, 1960, and 1981 (Chapman *et al.* 1994). The ICBTRT has characterized the spatial structure risk as “low” for the Wenatchee and Methow, “moderate” for the Entiat, and “high” for the Okanogan.

Critical habitat. Critical habitat was designated for this ESU on September 2, 2005 (70 FR 52488). It includes all Columbia River estuarine areas and river reaches upstream to Chief Joseph Dam and several tributary subbasins. The critical habitat designation for this DPS identifies PCEs that include sites necessary to support one or more life stages of steelhead. The DPS has 42 watersheds within its range. Three watersheds received a low rating, eight received a medium rating, and 31 rated a high conservation value to the DPS. In addition, the Columbia River rearing/migration corridor downstream of the spawning range was rated as a high conservation value. Limiting factors include: (1) Mainstem Columbia River hydropower system mortality, (2) reduced tributary streamflow, (3) tributary riparian degradation and loss of in-river wood, (4) altered tributary floodplain and channel morphology, and (5) excessive fine sediment and degraded tributary water quality.

Species-specific threats and limitations to recovery. The recovery of Upper Columbia River steelhead is impaired by a variety of anthropogenic factors, including impaired tributary fish passage, hatchery and harvest effects, completion, predation, disease, as well as fisheries management, including past introductions and persistence of non-native (exotic) fish species that continue to affect habitat conditions for listed species (Upper Columbia Salmon Recovery Board 2007, NMFS 2011i). Also significant is floodplain connectivity and function, channel structure and complexity, riparian areas and large woody debris recruitment, stream flow, and water quality have been degraded as a result of cumulative impacts of agriculture, forestry, and development.

Puget Sound steelhead

Distribution. Puget Sound steelhead occupy river basins of the Strait of Juan de Fuca, Puget Sound, and Hood Canal, Washington (Figure 20). 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.

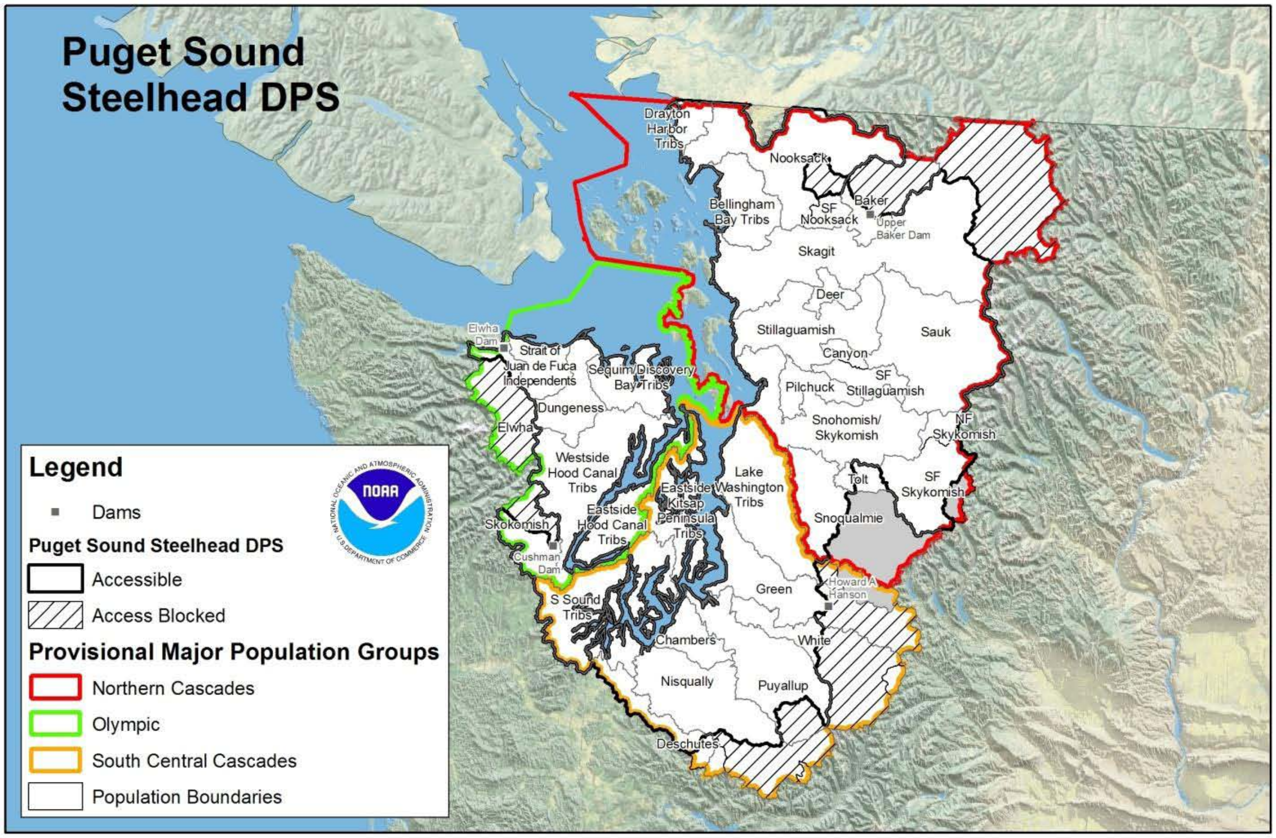

Figure 20. Population boundaries, dams, accessible areas, and extirpated reaches of Puget Sound steelhead

distribution, incorporating critical habitat boundaries (dark black line). Taken from NMFS (NMFS 2011f).

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.* (1996d), 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.* (1996d) report. Busby *et al.* (1996d) 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 (2011f) 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 24).

Table 24. 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	Unable to calculate

	(winter)	
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

Critical habitat. Critical habitat is not currently designated for Puget Sound steelhead. However, factors for essential habitat are under evaluation to designate future critical habitat.

Species-specific threats and limitations to recovery. Numerous impacts hinder the survival and recovery of Puget Sound steelhead (NMFS 2011f). 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.

Northern California steelhead

Distribution. Northern California steelhead includes steelhead in California coastal river basins from Redwood Creek south to the Gualala River, inclusive. Table 25 identifies populations within the Northern California Steelhead salmon ESU, their abundances, and hatchery input. Both winter and summer steelhead occur in this ESU, with immature half-pounders returning from the sea after only two to four months and overwintering in freshwater. These juveniles then outmigrate in the following spring.

Table 25. Northern California steelhead salmon populations, abundances, and hatchery contributions (Good *et al.* 2005b).

River	Historical abundance	Most recent spawner abundance	Hatchery abundance contributions
Redwood Creek	10,000	Unknown	Unknown
Mad River	6,000	162-384	Unknown
Eel River	82,000	3,127-21,903	Unknown
Mattole River	12,000	Unknown	Unknown
Ten Mile River	9,000	Unknown	Unknown
Noyo River	8,000	Unknown	Unknown
Big River	12,000	Unknown	Unknown
Navarro River	16,000	Unknown	Unknown
Garcia River	4,000	Unknown	Unknown
Gualala River	16,000	Unknown	Unknown
Other Humboldt County streams	3,000	Unknown	Unknown

River	Historical abundance	Most recent spawner abundance	Hatchery abundance contributions
Other Mendocino County streams	20,000	Unknown	Unknown
Total	**198,000**	**Unknown**	

Status and trends. Northern California steelhead were listed as threatened on June 7, 2000 (65 FR 36074), and when their status was reviewed on January 5, 2006 they retained that classification (71 FR 834). Long-term data sets are limited. Before 1960, estimates of abundance were available from dam counts in the upper Eel River (Cape Horn Dam–annual average number of adults was 4,400 in the 1930s), the South Fork Eel River (Benbow Dam–annual average number of adults was 19,000 in the 1940s), and the Mad River (Sweasey Dam–annual average number of adults was 3,800 in the 1940s). Estimates of steelhead spawning populations for many rivers in this DPS totaled 198,000 by the mid-1960s. At the time of the first status review, adult escapement trends could be computed on seven populations. Five of the seven populations exhibited declines, while two exhibited increases, with a range of almost 6% annual decline to a 3.5% increase. At the time, little information was available on the actual contribution of hatchery fish to natural spawning (Busby *et al.* 1996d).

More recent time serious data come from snorkel counts conducted on summer-run steelhead in the Middle Fore Eel River. An estimate of lambda over the interval 1966-2002 was made and a random-walk with drift model fitted using Bayesian assumptions. Good *et al.* (2005b) estimated lambda at 0.98 with a 95% confidence interval of 0.93 and 1.04. The result is an overall downward trend in both the long and short term. Juvenile data were also recently examined. Both upward and downward trends were apparent (Good *et al.* 2005b).

Critical habitat. Critical habitat was designated for this species on September 2, 2005 (70 FR 52488). The critical habitat designation for this DPS identifies PCEs that include sites necessary to support one or more life stages of steelhead.

Snake River steelhead

Distribution. Snake River basin steelhead are an inland species that occupy the Snake River basin of southeast Washington, northeast Oregon, and Idaho (Figure 21). The historic spawning range of this species included the Salmon, Pahsimeroi, Lemhi, Selway, Clearwater, Wallowa, Grande Ronde, Imnaha, and Tucannon Rivers. Table 26 identifies populations within ESU, their abundances, and hatchery input.

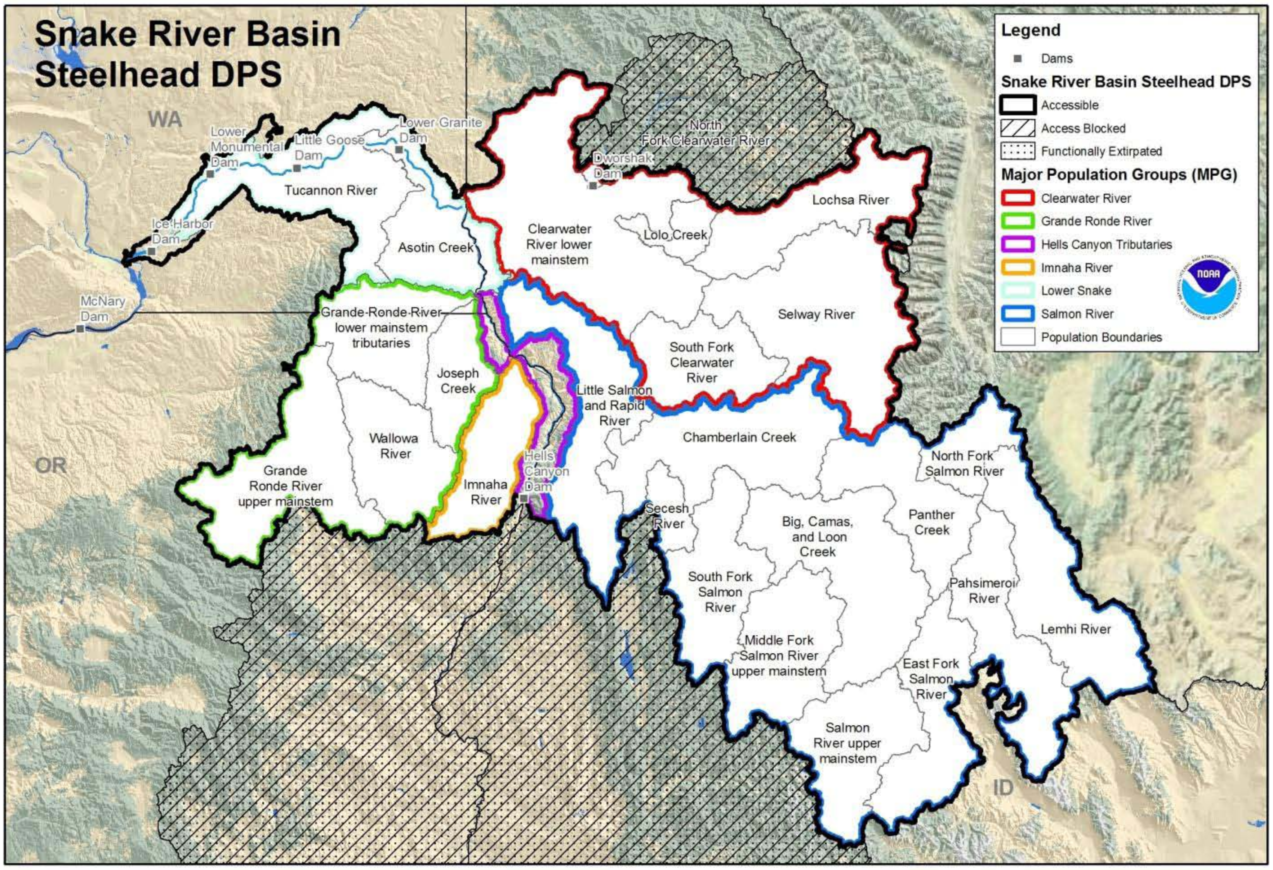

Figure 21. Population boundaries, dams, accessible areas, and extirpated reaches, of Snake River steelhead distribution, incorporating critical habitat boundaries (dark black line). Taken from NMFS (2011l).

Snake River steelhead occupy a basin that is annually warmer and drier than other steelhead DPSs. Snake River Basin steelhead are generally classified as summer run, based on their adult run timing pattern. Sexually immature adults enter the Columbia River from late June to October as A-run fish that spend one year in the ocean, or larger B-run fish that spend two years at sea. Adults typically migrate upriver until they reach tributaries from 3,300-6,600 feet above sea level where they spawn between March and May of the following year. Adult spawner survival is much higher for individuals returning in early to mid May than for those returning in mid June (Scheuerell *et al.* 2009). Unlike other anadromous salmonids, some adult steelhead survive spawning, return to the sea, and later return to spawn a second time. After hatching, juvenile Snake River steelhead typically spend two to three years in river before they smolt and migrate to the ocean.

Table 26. Snake River basin steelhead salmon populations, abundances, and hatchery contributions (Good *et al.* 2005b) [rpm = redds per mile]).

River	Historical abundance	Most recent spawner abundance	Hatchery abundance contributions
Tucannon River	3,000	257-628	26%
Lower Granite run	Unknown	70,721-259,145	86%
Snake A run	Unknown	50,974-25,950	85%
Snake B run	Unknown	9,736-33,195	89%
Asotin Creek	Unknown	0-543 redds	Unknown

River	Historical abundance	Most recent spawner abundance	Hatchery abundance contributions
Upper Grande Ronde River	15,000	1.54 rpm	23%
Joseph Creek	Unknown	1,077-2,385	0%
Imnaha River	4,000	3.7 rpm	20%
Camp Creek	Unknown	55-307	0%
Total	**22,000 (min)**		

Managers classify summer steelhead runs into two groups based on ocean age and adult size when they return to the Columbia River. A-run steelhead are predominately one-year-old ocean fish. B-run steelhead are predominately two-year-old ocean fish. A-run populations are found in lower Clearwater River, upper Salmon River, lower Salmon River and their tributaries, the Grand Ronde River, Imnaha River, and possibly the Snake River's mainstem tributaries below Hells Canyon Dam. B-run steelhead occupy four major subbasins not occupied by A-run steelhead: two on the Clearwater River (Lochsa and Selway) and two on the Salmon River (Middle Fork and South Fork Salmon). Some natural B-run steelhead spawn in the mainstem section of the Clearwater River and its major tributaries. There are alternative escapement objectives for B-run steelhead. B-run steelhead represent at least one-third and as much as three-fifths of the production capacity of the DPS (10,000 from the Columbia River Fisheries Management Plan and 31,400 from Idaho).

Status and trends. Snake River steelhead were listed as threatened in 1997 (62 FR 43937), when their status was reviewed on January 5, 2006 they retained that classification (71 FR 834). The ICBTRT (2003) identified 23 populations in the following six major population groups in this species: Clearwater River, Grande Ronde River, Hells Canyon, Imnaha River, Lower Snake River, and Salmon River. Snake River Basin steelhead remain spatially well distributed in each of the six major geographic areas in the Snake River basin (Good *et al.* 2005b). Environmental conditions are generally drier and warmer in these areas than in areas occupied by other steelhead species in the Pacific northwest. Snake River Basin steelhead were blocked from portions of the upper Snake River beginning in the late 1800s and culminating with the construction of Hells Canyon Dam in the 1960s. The Snake River Basin steelhead "B run" population levels remain particularly depressed.

The paucity of information on adult spawning escapement for specific tributary production areas for Snake River Basin steelhead made a quantitative assessment of viability difficult. Annual return estimates are limited to counts of the aggregate return over Lower Granite Dam, and spawner estimates for the Tucannon, Grande Ronde, and Imnaha Rivers. The 2001 return over Lower Granite Dam was substantially higher relative to the low levels seen in the 1990s; the recent 5-year mean abundance (14,768 natural returns) was approximately 28% of the interim recovery target level (52,000 natural spawners). The 10-year average for natural-origin steelhead passing Lower Granite Dam between 1996 and 2005 was 28,303 adults. Parr densities in natural production areas, which are another indicator of population status, have been substantially below estimated capacity for several decades. The Snake River supports approximately 63% of the total natural-origin production of steelhead in the Columbia River Basin. Genetic diversity is currently being depressed by the displacement of natural fish by hatchery fish (declining proportion of natural-origin spawners). Homogenization of hatchery stocks occur within basins, and some stocks exhibit high stray rates. Little change from the 2005 review was found in the

2011 status review (NMFS 2011l).

Critical habitat. Critical habitat was designated for this species on September 2, 2005 (70 FR 52488). The critical habitat designation for this ESU identifies PCEs that include sites necessary to support one or more life stages of steelhead. The critical habitat designation for this ESU identifies PCEs that include sites necessary to support one or more steelhead life stages. Of the 291 fifth order streams reviewed in this DPS, 220 were rated as high, 44 were rated as medium, and 27 were rated as low conservation value. Limiting factors identified for Snake River Basin steelhead include: (1) hydrosystem mortality, (2) reduced stream flow, (3) altered channel morphology and floodplain, (4) excessive sediment, (5) degraded water quality, (6) harvest impacts, and (7) hatchery impacts (NMFS 2006i).

Species-specific threats and limitations to recovery. Snake River Basin steelhead recovery is limited by predation, impaired water quality and increased temperature, harvest (particularly of B-run steelhead), genetic diversity stemming from relerase of out-of-population hatchery fish, impaired tributary fish passage (ICTRT 2006, NMFS 2011l). Also significant is floodplain connectivity and function, channel structure and complexity, riparian areas and large woody debris recruitment, stream flow, and water quality have been degraded as a result of cumulative impacts of agriculture, forestry, and development.

Upper Willamette River steelhead

Distribution. Upper Willamette River steelhead occupy the Willamette River and its tributaries upstream of Willamette Falls (Figure 22). Table 27 identifies the four populations within the Upper Willamette River Steelhead salmon ESU, their abundances, and hatchery input.

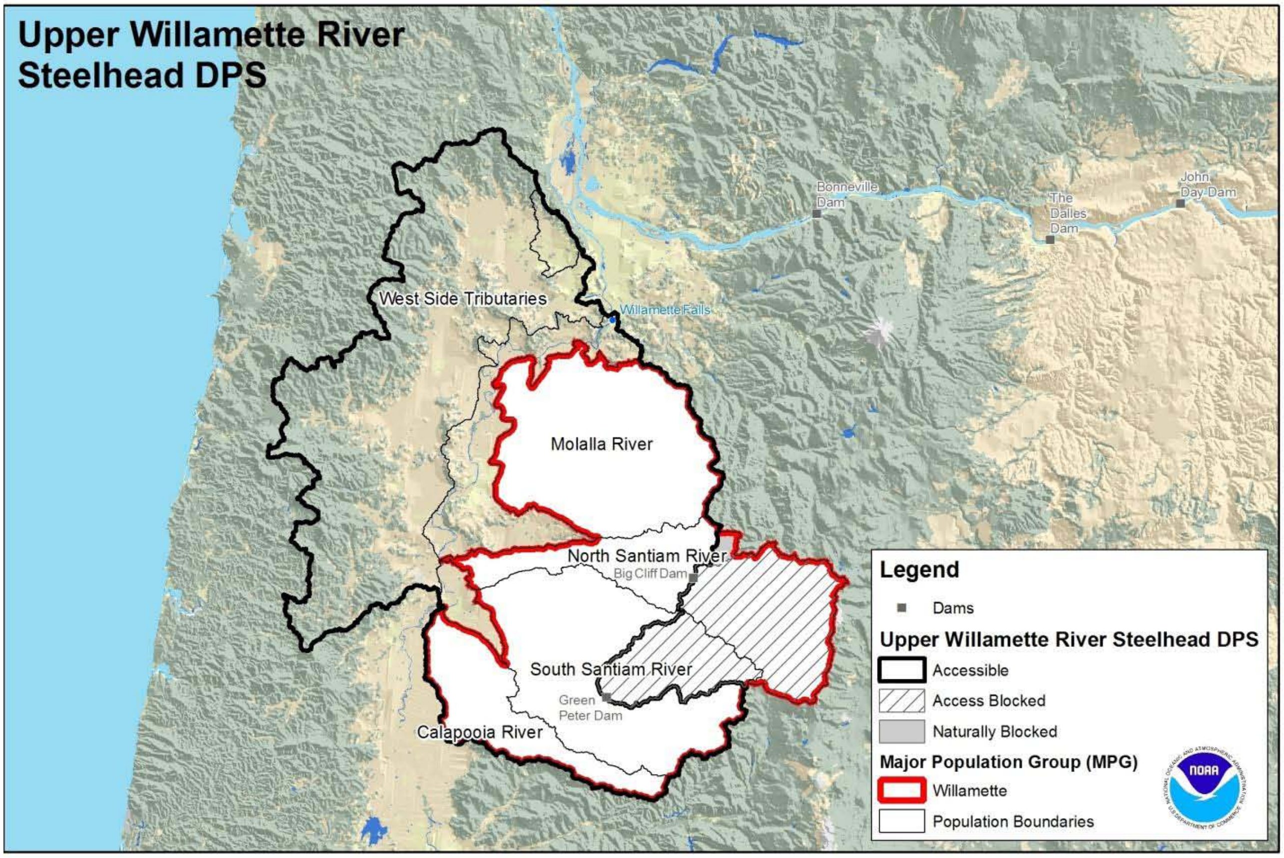

Figure 22. Population boundaries, dams, accessible areas, and extirpated reaches, of upper Willamette River steelhead distribution, incorporating critical habitat boundaries (dark black line). Taken from NMFS (2011).

Table 27. Upper Willamette river steelhead populations, abundances, and hatchery contributions (Good *et al.* 2005b) [rpm = redds per mile]).

Population	**Historical abundance**	**Most recent spawner abundance**	**Hatchery abundance contributions**
Mollala Rivers	Unknown	0.972 rpm	Unknown
North Santiam River	Unknown	0.963 rpm	Unknown
South Santiam River	Unknown	0.917 rpm	Unknown
Calapooia River	Unknown	1.053 rpm	Unknown
Total	**Unknown**	**5,819**	

This is a late-migrating winter group that enters freshwater in January and February (Howell *et al.* 1985a). They do not ascend to their spawning areas until late March or April (Dimick and Merryfield 1945) and spawning occurs from April to June. The smolt migration past Willamette Falls also begins in early April and proceeds into early June, peaking in early- to mid-May (Howell *et al.* 1985a). Smolts generally migrate through the Columbia via Multnomah Channel rather than the mouth of the Willamette River. Most spend two years in the ocean before re-entering natal rivers to spawn (Busby *et al.* 1996b). Steelhead in the Upper Willamette River DPS generally spawn once or twice, although some may spawn three times. Repeat spawners are predominantly female and generally account for less than 10% of the total run size (Busby *et al.* 1996b).

Status and trends. Upper Willamette River steelhead were listed as threatened in 1999 (64 FR

14517) and when their status was reviewed on January 5, 2006 they retained that classification (71 FR 834). A major threat to Upper Willamette River steelhead results from artificial production practices. Fishways built at Willamette Falls in 1885 have allowed Skamania-stock summer steelhead and early-migrating winter steelhead of Big Creek stock to enter the range of Upper Willamette River steelhead. The population of summer steelhead is almost entirely maintained by hatchery salmon, although natural-origin, Big Creek-stock winter steelhead occur in the basin (Howell *et al.* 1985a). In recent years, releases of winter steelhead are primarily of native stock from the Santiam River system.

Steelhead in this DPS are depressed from historical levels (McElhaney *et al.* 2007). Although long-term growth is negative, short term abundance studies support a recent upturn in population numbers until 2002, when the population peaked and has since returned to historical levels (2008 abundance was 4,915 individuals passing Willamette Falls and 2,110 in 2009)(McElhaney *et al.* 2007, NMFS 2011j). Spatial structure for the North and South Santiam populations has been substantially reduced by the loss of access to the upper North Santiam basin and the Quartzville Creek watershed in the South Santiam subbasin due to dam construction lacking passage facilities (McElhaney *et al.* 2007). Additionally, habitat in the Molalla subbasin has been reduced significantly by habitat degradation and in the Calapooia by habitat degradation as well as passage barriers. Finally, the diversity of some populations has been eroded by small population size, the loss of access to historical habitat, legacy effects of past winter-run hatchery releases, and the ongoing release of summer steelhead (McElhaney *et al.* 2007).

Critical habitat. Critical habitat was designated for this species on September 2, 2005 (70 FR 52488). The designation includes all Columbia River estuarine areas and river reaches upstream to the confluence with the Willamette River as well as specific steam reaches in the following subbasins: Upper Willamette, North Santiam, South Santiam, Middle Willamette, Molalla/Pudding, Yamhill, Tualatin, and Lower Willamette (NMFS 2005b). The critical habitat designation for this DPS identifies PCEs that include sites necessary to support one or more life stages of steelhead. Of 43 subbasins reviewed in NMFS' assessment of critical habitat for Upper Willamette River steelhead, 20 subbasins were rated as having a high conservation value, while six were rated as having a medium value and 17 were rated as having a low value to the conservation of the DPS.

Land management activities have severely degraded stream habitat conditions in the Willamette River mainstem above Willamette Falls and associated subbasins. In the Willamette River mainstem and lower sub-basin mainstem reaches, high density urban development and widespread agricultural effects have reduced aquatic and riparian habitat quality and complexity, and altered sediment and water quality and quantity, and watershed processes. The Willamette River, once a highly braided river system, has been dramatically simplified through channelization, dredging, and other activities that have reduced rearing habitat by as much as 75%. In addition, the construction of 37 dams in the basin blocked access to more than 435 miles of stream and river spawning habitat. The dams alter the temperature regime of the Willamette River and its tributaries, affecting the timing and development of naturally-spawned eggs and fry. Agriculture, urbanization, and gravel mining on the valley floor logging in the Cascade and Coast Ranges contribute to increased erosion and sediment loads throughout the basin.

The mainstem Willamette River has been channelized and stripped of large wood. Development began to encroach on the riparian forest beginning in the 1870s (Sedell and Froggatt 1984).

Gregory *et al.* (2002d) calculated that the total mainstem Willamette River channel area decreased from 41,000 to 23,000 acres between 1895 and 1995. They noted that the lower reach, from the mouth of the river to Newberg (RM 50), is confined within a basaltic trench, and that due to this geomorphic constraint, less channel area has been lost than in upstream areas. The middle reach from Newberg to Albany (RM 50 to 120) incurred losses of 12% primary channel area, 16% side channels, 33% alcoves, and 9% islands. Even greater changes occurred in the upper reach, from Albany to Eugene (RM 187). There, approximately 40% of both channel length and channel area were lost, along with 21% of the primary channel, 41% of side channels, 74% of alcoves, and 80% of island areas.

The banks of the Willamette River have more than 96 miles of revetments; approximately half were constructed by the ACOE. Generally, the revetments were placed in the vicinity of roads or on the outside bank of river bends, so that while only 26% of the total length is revetted, 65% of the meander bends are revetted (Gregory *et al.* 2002c). The majority of dynamic sections have been armored, reducing adjustments in channel bed and sediment storage by the river, and thereby diminishing both the complexity and productivity of aquatic habitats (Gregory *et al.* 2002b).

Riparian forests have diminished considerably in the lower reaches of the Willamette River (Gregory *et al.* 2002a). Sedell and Froggatt (1984) noted that agriculture and cutting of streamside trees were major agents of change for riparian vegetation, along with snagging of large wood in the channel. The reduced shoreline, fewer and smaller snags, and reduced riparian forest comprise large functional losses to the river, reducing structural features, organic inputs from litter fall, entrained allochthonous materials, and flood flow filtering capacity. Extensive changes began before the major dams were built, with navigational and agricultural demands dominating the early use of the river. The once expansive forests of the Willamette River floodplain provided valuable nutrients and organic matter during flood pulses, food sources for macroinvertebrates, and slow-water refugia for fish during flood events. These forests also cooled river temperatures as the river flowed through its many channels.

Gregory *et al.* (2002a) described the changes in riparian vegetation in river reaches from the mouth to Newberg, from Newberg to Albany, and from Albany to Eugene. They noted that the riparian forests were formerly a mosaic of brush, marsh, and ash tree openings maintained by annual flood inundation. Below the City of Newberg, the most noticeable change was that conifers were almost eliminated. Above Newberg, the formerly hardwood-dominated riparian forests along with mixed forest made up less than half of the riparian vegetation by 1990, while agriculture dominated. This conversion has reduced river shading and the potential for recruitment of wood to the river, reducing channel complexity and the quality of rearing, migration and spawning habitats

Hyporheic flow in the Willamette River has been examined through discharge measurements and found to be significant in some areas, particularly those with gravel deposits (Wentz *et al.* 1998, Fernald *et al.* 2001). The loss of channel complexity and meandering that fosters creations of gravel deposits decreases the potential for hyporheic flows, as does gravel mining. Hyporheic flow processes water and affects its quality on reemerging into the main channel, stabilizing variations in physical and chemical water characteristics. Hyporheic flow is important for ecological functions, some aspects of water quality (such as temperature and dissolved oxygen), and some benthic invertebrate life stages. Alcove habitat, which has been limited by channelization, combines low hydraulic stress and high food availability with the potential for hyporheic flows across the steep hydraulic gradients in the gravel separating them from the main channel (Fernald *et al.* 2001).

Species-specific threats and limitations to recovery. Habitat degredation and hatchery-related impacts are significant in hindering the recovery of Upper Willamette River steelhead (Guimarães *et al.* 2011, ODFW and NMFS 2011). Floodplain connectivity and function, channel structure and complexity, riparian areas and large wood recruitment, and stream flow have been degraded as a result of cumulative impacts of agriculture, forestry, and development. Degraded water quality and altered temperature as a result of both tributary dams and the cumulative impacts of agriculture, forestry, and urban development also has reduced the quality of available habitat. Spawning individuals have reduced access to spawning and rearing habitats mainly as a result of artificial barriers in spawning tributaries. Genetic effects of non-native summer steelhead hatchery programs and anthropogenic introductions of non-native species and out-of-ESU races of salmon or steelhead have increased predation and competition on native UWR steelhead.

South-Central California coast steelhead

Distribution. The South-Central California steelhead DPS includes all naturally spawned populations of steelhead (and their progeny) in streams from the Pajaro River (inclusive) to, but not including the Santa Maria River, California. Only winter steelhead are found in this ESU. Migration and spawn timing are similar to adjacent steelhead populations.

Status and trends. South-Central California Coast steelhead were listed as threatened in 1997, when their status was reviewed on January 5, 2006 they retained that classification (71 FR 834). Historical data on the South-Central California Coast steelhead DPS are limited. In the mid-1960s the California Department of Fish and Game estimated that the adult population at about 18,000. We know of no recent estimates of the total DPS. However, five river systems, the Pajaro, Salinas, Carmel, Little Sur, and Big Sur, indicate that runs are currently less than 500 adults. Past estimates for these basins were almost 5,000 fish. Carmel River time series data indicate that the population declined by about 22% per year between 1963 and 1993 (Good *et al.* 2005b). From 1991 the population increased from one adult, to 775 adults at San Clemente Dam. Good *et al.* (2005b) thought that this recent increase seemed too great to attribute simply to improved reproduction and survival of the local steelhead population. Other possibilities were considered including that the substantial immigration or transplantation occurred, or that resident trout production increased as a result of improved environmental conditions within the basin.

Critical habitat. Critical habitat was designated for this species on September 2, 2005 (70 FR 52488). The critical habitat designation for this DPS identifies PCEs that include sites necessary to support one or more life stages of steelhead. The critical habitat designation for this DPS identifies PCEs that include sites necessary to support one or more steelhead life stages.

Southern California Steelhead

Distribution. Southern California steelhead occupy rivers from the Santa Maria River to the US –Mexico border. Table 28 identifies populations within the Southern California Steelhead salmon ESU, their abundances, and hatchery input. Migration and life history patterns of southern California steelhead are dependent on rainfall and streamflow (Moore 1980). Steelhead within this ESU can withstand higher temperatures than more northerly populations and the relatively warm and productive waters of the Ventura River permit more rapid juvenile growth (Moore 1980).

Table 28. Southern California steelhead populations, abundances, and hatchery contributions (Good *et al.* 2005).

River	Historical Abundance	Most Recent Spawner Abundance	Hatchery Abundance Contributions
Santa Ynez River	12,995-30,000	Unknown	Unknown
Ventura River	4,000-6,000	Unknown	Unknown
Matilija River	2,000-2,500	Unknown	Unknown
Creek River	Unknown	Unknown	Unknown
Santa Clara River	7,000-9,000	Unknown	Unknown
Total	**32,000-46,000**	**<500**	

Status and trends. Southern California steelhead were listed as endangered in 1997 (62 FR 43937), when their status was reviewed on January 5, 2006 they retained that classification (71 FR 834). In many watersheds throughout Southern California, dams isolate steelhead from historical spawning and rearing habitats and alter the hydrology of the basin (e.g., Twitchell Reservoir within the Santa Maria River watershed, Bradbury Dam within the Santa Ynez River watershed, Matilija and Casitas dams within the Ventura River watershed, Rindge Dam within the Malibu Creek watershed). Based on combined estimates for the Santa Ynez, Ventura, and Santa Clara rivers, and Malibu Creek, an estimated 32,000 to 46,000 adult steelhead occupied this DPS. In contrast, less than 500 adults are estimated to occupy the same four waterways presently. The last estimated run size for steelhead in the Ventura River, which has its headwaters in Los Padres National Forest, is 200 adults (Busby *et al.* 1996c, Busby *et al.* 1996d).

Critical habitat. Critical habitat was designated for this species on September 2, 2005. The designation identifies PCEs that include sites necessary to support one or more steelhead life stages and, in turn, these sites contain the physical or biological features essential for the species conservation.

Southern green sturgeon

Description of the species. The species is divided into two genetically distinct but physically indistinguishable clades: a Northern DPS whose populations are relatively healthy, and a Southern DPS that has undergone significant decline (Adams *et al.* 2007). Only the Southern DPS of green sturgeon is listed under the ESA.

Distribution. Green sturgeon occur along the west coast of North America from Mexico to the Bering Sea (Adams *et al.* 2002, Moyle 2002a, Colway and Stevenson 2007), as well in freshwater rivers and estuaries in between. Both northern DPS and southern DPS green sturgeon occupy coastal estuaries and coastal marine waters from southern California to Alaska, including Humbolt Bay, the lower Columbia River estuary, Willapa Bay, Grays Harbor and southeast

Alaska. In general, green sturgeon are more common north of Point Conception, California. This DPS occupies a limited number of river systems, primarily the main stems and tributaries of the Sacramento, Feather, and San Joaquin rivers, all of which drain to San Francisco Bay (Israel *et al.* 2009). The Sacramento River is the only known spawning habitat for the Southern green sturgeon DPS (Moyle *et al.* 1992a, CDFG 2002). Spawning in the San Joaquin River does not presently occur, likely as a result of damming and irrigation development. Adult and juvenile green sturgeon occur throughout the Sacramento River. Although Southern DPS individuals in freshwater occur primarily in the aforementioned waterways, some have been captured in rivers of the Northern DPS, including the Klamath, Rogue, and Columbia rivers of northern California and Oregon. Individuals have also rarely been captured in Southern California waters, but their abundance increases north of Point Conception.

Upon outmigration from fresh water, subadult green sturgeon disperse widely along through continental shelf waters of the west coast within the 110 meter contour (NMFS and USFWS 2005).(Moyle *et al.* 1992a, Erickson and Hightower 2007). Captures provide the limited knowledge available on green sturgeon distribution and habitat in estuaries and the North Pacific Ocean. Adults are known from coastal estuaries and bays, including Willapa and Nehalem bays, Grays Harbor, and the Columbia River estuary (EPIC *et al.* 2001). It appears that Southern DPS green sturgeon preferentially distribute north of the river mouth from whence they emerge as juveniles during fall and move into bays and estuaries during summer and fall (Moser and Lindley 2007, Israel *et al.* 2009). Spawning individuals move south during spring to natal freshwater habitat (Lindley *et al.* 2008).

Reproduction. Adults infrequently occupy freshwater reaches primarily to spawn and spend the majority of their lives in estuaries and coastal marine waters (Wilson and McKinley 2004). Sturgeon, like salmon, possess strong site fidelity and will return to their natal streams to spawn (Bemis and Canard 1997). This occurs between late February and July, with a peak in mid-April to mid-June, hold in deep pools and return to salt water in the fall early, often with the first increases in fall flows, although Northern DPS individuals in the Rogue River seek out slow-flow and off-channel coves (Moyle *et al.* 1992a, Moyle *et al.* 1995a, Erickson *et al.* 2001b, Rien *et al.* 2001a, Erickson and Webb 2007a, Heublein *et al.* 2009). Fish then tend to aggregate in deep pools, where they will over-summer before outmigrating in the fall, although some fish have been observed outmigrating relatively soon after presumed spawning events (Heublein *et al.* 2009). However, multiple spawning events may occur annually, unlike annual spawning of salmonids (Heublein *et al.* 2009). Adults may spend over 6 months in freshwater until water temperatures drop below 50º F in fall or winter or when a significant flow even occurs, although temperature or flow cues are not always needed to initiate downstream migration (Erickson and Webb 2007a, Heublein *et al.* 2009). In the Sacramento River adult green sturgeon spawn in late spring and early summer above Hamilton City, above Red Bluff Diversion Dam, and possibly as far upstream as Keswick Dam (CDFG 2002, Heublein *et al.* 2009). It appears that specific habitat for spawning includes large cobblestones (where eggs can settle between), although spawning is known to occur over clean sand or bedrock.

Adults likely return on a 2 to 5 year basis for spawning starting at 15 to 30 years of age for males and 17 to 40 for females (Adams *et al.* 2002, Moyle 2002a, Van Eenennaam *et al.* 2006). Most male spawners are young (17 to 18 years) while females on the spawning grounds are often older

(27 to 28 years). Females produce roughly 60,000 to 140,000 eggs per spawning event (Scott and Crossman 1973c, Moyle *et al.* 1992a). Each egg is large by sturgeon standards so as to contain greater nourishment for embryos (Cech Jr. *et al.* 2000). Temperature may trigger spawning behavior, with ranges of 48º to 62º F being influential (Moyle *et al.* 1995a). Water temperature is also critical for egg survival with temperatures above 68º F being fatal to developing embryos (Cech Jr. *et al.* 2000).

Growth. Green sturgeon spend their first 1 to 4 years in their natal streams and rivers (Nakamoto *et al.* 1995b, Beamesderfer and Webb 2002), although they are believed to be physiologically adapted to sea water survival at 6 months of age (Allen and Cech 2007, Allen and J.J. Cech 2007, Allen *et al.* 2009). Larvae are active at night, a behavior that likely reduces predation and avoids being moved downstream more than necessary (Cech Jr. *et al.* 2000). Green sturgeon larvae grow very rapidly, reaching about 300 mm by age one (Deng 2000). Growth of the larval stage is ideal at 59º F, reduced below 52º F and above 66º F, and highly reduced above 75ºF (Cech Jr. *et al.* 2000). Juvenile green sturgeon of the Northern DPS (Rogue River) have been captured in the Rogue River estuary from April until the end of November (Rien *et al.* 2001a). Green sturgeon are a long-lived fish, and likely live for 60 to 70 years (Moyle 2002a).

Feeding. While in fresh water, juveniles feed on a variety of fishes and invertebrates (Moyle *et al.* 1992a). One juvenile from the Sacramento-San Joaquin estuary was found to have preyed most commonly upon opisthobranch mollusks (*Philline* sp.), with bay shrimp (*Crangon* sp.) and overbite clams (*Potamocorbula amurensis*) as secondary prey. Other juveniles in the Sacramento River delta feed on opossum shrimp (*Neomysis mercedis*) and *Corophium* amphipods (Radtke 1966).

The limited feeding data available for adult green sturgeon show that they consume benthic invertebrates including shrimp, clams, chironomids, copepods, mollusks, amphipods, and small fish (Houston 1988, Moyle *et al.* 1992a)(Houston 1988; Moyle *et al.* 1992; Wilson and McKinley 2004; Dumbauld *et al.* 2008)(Moyle *et al.* 1992a, Wilson and McKinley 2004, Dumbauld *et al.* 2008). Starting as larvae, sturgeon use electroreception to identify prey. Olfaction and taste may also be important to foraging, while vision is thought play a minor role in prey capture (Miller 2004).

Status and trends. NMFS listed the southern population of the North American green sturgeon as threatened on April 7, 2006 (71 FR 17757). Trend data for green sturgeon is severely limited. Available information comes from two predominant sources, fisheries and tagging. Only three data sets were considered useful for the population time series analyses by NMFS' biological review team: the Klamath Yurok Tribal fishery catch, a San Pablo sport fishery tag returns, and Columbia River commercial landings (NMFS and USFWS 2005). Using San Pablo sport fishery tag recovery data, the California Department of Fish and Game produced a population time series estimate for the southern DPS. San Pablo data suggest that green sturgeon abundance may be increasing, but the data showed no significant trend. The data set is not particularly convincing, however, as it suffers from inconsistent effort and since it is unclear whether summer concentrations of green sturgeon provide a strong indicator of population performance (NMFS and USFWS 2005). Although there is not sufficient information available to estimate the current population size of southern green sturgeon, catch of juveniles during state and federal salvage operations in the Sacramento delta are low in comparison to catch levels before the mid-1980s.

Natural threats. Green sturgeon eggs and larvae are likely preyed upon by a variety of larger fish and animals, while sub-adult and adult sturgeon may occasionally be preyed upon by shark sea lions, or other large body predators. Physical barriers, changes in water flow and temperatures may also affect fresh water survival.

Anthropogenic threats. The principle threat to southern green sturgeon comes from a drastic reduction in available spawning area from impassible barriers (e.g., Oroville, Shasta and Keswick dams). Other threatens include potentially lethal temperature limits, harvest, entrainment by water projects and toxins and invasive species (Adams *et al.* 2007, Erickson and Webb 2007b, Lackey 2009). Since this DPS is composed of a single spawning population within the Sacramento River, stochastic variation in environmental conditions and significant fluctuations in demographic rates increases the risk of extinction for this DPS.

Climate change has the potential to affect sturgeon in similar, if not more significant ways it affects salmonids. Elevated air temperatures could lead to precipitation falling as rain instead of snow. Additionally, snow would likely melt sooner and more rapidly, potentially leading to greater flooding during melting and lower water levels at other times, as well as warmer river temperatures. Although sturgeon can spawn over varied benthic habitat, they prefer localized depressions in riverbeds (Moyle *et al.* 1992b, Moyle *et al.* 1995b, Erickson *et al.* 2001a, Rien *et al.* 2001b). Increased extremes in river flow (i.e., periods of flooding and low flow) can alternatively disrupt and fill in spawning habitat that sturgeon rely upon (ISAB 2007b). If water flow is low during migration events, it is likely that new obstacles can impede or block sturgeon movement. As with other anadromous fishes, sturgeon are uniquely evolved to the environments that they live in. Because of this specificity, broad scale changes in environment can be difficult to adapt to, including changes in water temperature (Cech *et al.* 2000). Sturgeon are also sensitive to elevated water temperatures. Temperature triggers spawning behavior. Warmer water temperatures can initial spawning earlier in a season for salmon and the same can be true for sturgeon (ISAB 2007b). If river and lake temperatures become anomalously warm, juvenile sturgeon may experience elevated mortality due to lack of cooler water refuges in freshwater habitats. Apart from direct changes to sturgeon survival, altered water temperatures may disrupt habitat, including the availability of prey (ISAB 2007b). Warmer temperatures may also have the effect of increasing water use in agriculture, both for existing fields and the establishment of new ones in once unprofitable areas (ISAB 2007b). This means that streams, rivers, and lakes will experience additional withdrawal of water for irrigation and increasing contaminant loads from returning effluent. Overall, it is likely that global warming will increase pressures on sturgeon survival and recovery.

Studies from other sturgeon species indicate that sturgeon readily bioaccumulate contaminants. White sturgeon from the Kootenai River have been found to contain aluminum, arsenic, cadmium, chromium, cobalt, copper, iron, lead, manganese, mercury, nickel, selenium, zinc, DDE, DDT, PCBs, and other organochlorines (Kruse and Scarnecchia 2001). Mercury has also been identified from white sturgeon of the lower Columbia River (Webb *et al.* 2006). Numerous organochlorines, including DDT, DDD, DDE, chlordane, and dieldrin have also been identified in these fish (Foster *et al.* 2001). Observed concentrations are likely sufficient to influence reproductive physiology.

Critical habitat. On October 9, 2009, NMFS designated critical habitat for southern green sturgeon (74 FR 52300). The geographical area identified as critical habitat is based upon the overlapping distribution of the southern and northern DPS, and encompasses all areas where the presence of southern green sturgeon have been confirmed or where their presence is likely. Therefore the geographical area defined as critical habitat is the entire range of the biological species, green sturgeon, from the Bering Sea, AK, to Ensenada, Mexico. Specific fresh water areas include the Sacramento River, Feather River, Yuba River, and the Sacramento-San Joaquin Delta. Specific coastal bays and estuaries include estuaries from Elkhorn Slough, California, to Puget Sound, Washington. Coastal marine areas include waters along the entire biological species range within a depth of 60 fathoms. The principle biological or physical constituent elements essential for the conservation of southern green sturgeon in fresh water include: food resources; substrate of sufficient type and size to support viable egg and larval development; water flow, water quality such that the chemical characteristics support normal behavior, growth and viability; migratory corridors; water depth; and sediment quality. Primary constituent elements of estuarine habitat include food resources, water flow, water quality, migratory corridors, water depth, and sediment quality. The specific primary constituent elements of marine habitat include food resources, water quality, and migratory corridors.

Critical habitat of the Southern DPS of green sturgeon is threatened by several anthropogenic factors. Four dams and several other structures currently are impassible for green sturgeon to pass on the Sacramento, Feather, and San Joaquin rivers, preventing movement into spawning habitat. Threats to these riverine habitats also include increasing temperature, insufficient flow that may impair recruitment, the introduction of striped bass that may eat young sturgeon and compete for prey, and the presence of heavy metals and contaminants in the river. The application of pesticides may adversely affect prey resources and water quality within the bays and estuaries, as well as the growth and reproductive health of Southern DPS green sturgeon through bioaccumulation. Other activities of concern include those that may disturb bottom substrates, adversely affect prey resources, or degrade water quality through re-suspension of contaminated sediments. Of particular concern are activities that affect prey resources. Prey resources can be affected by: commercial shipping and activities generating point source pollution and non-point source pollution that can discharge contaminants and result in bioaccumulation of contaminants in green sturgeon; disposal of dredged materials that can bury prey resources; and bottom trawl fisheries that can disturb the bottom (but may result in beneficial or adverse effects on prey resources for green sturgeon). In addition, petroleum spills from commercial shipping activities and proposed alternative energy hydrokinetic projects may affect water quality or hinder the migration of green sturgeon along the coast (USDC 2009b).

Shortnose sturgeon

Description of the species. Shortnose sturgeon occur along the Atlantic Coast of North America, from the St. John River in Canada, south to the St. John's River in Florida. NMFS' recovery plan (1998b) recognized 19 wild populations based on their strong fidelity to their natal streams (Table 29), and several captive populations (from a Savannah River broodstock) that are maintained for educational and research purposes (NMFS 1998b). Although these populations are geographically isolated, genetic analyses suggest that individual shortnose sturgeon move between some of these populations each generation (Quattro 2002, Wirgin *et al.* 2005a).

Table 29. Shortnose sturgeon populations and their estimated abundances.

Population (Location)[a]	Data Series	Abundance Estimate (C.I.)[b]	Population Segment	Reference
Saint John River (Canada)	1973-1977	18,000 (+/-30%)	Adults	Dadswell 1979
Kennebecasis River (Canada)	1998-2005	2,068 (801-11,277)		COSEWIC 2005
Kennebecasis River	2005	4,836 (+/-69)		Li *et al.* 2007, NMFS unpubl.
Penobscot River (ME)	2006-2007	1,049 (673-6,939)		UME 2008
	2008	1739 (846-3653)	Summer	P. Dionne, pers. comm..
		667 (451-1013)	Fall	P. Dionne, pers. comm..
Kennebec River (ME)	1977-1981	7,222 (5,046-10,765)	Adult	Squiers *et al.* 1982
	2003	9,488 (6,942-13,358)	Adults	Squiers 2003
Merrimack River (MA)	1987-1991	32 (20-79)	Adults	Kynard & Kieffer, unpubl.; NMFS unpubl.
Connecticut River (MA, CT)	1989-2002	1,042-1,580 [c]	Adults	Savoy 2004
Upper Connecticut River[d]	1976-1977	516 (317–898)	Total	Taubert 1980; NMFS 1998a
	1977-1978	370 (235–623)	Total	Taubert 1980; NMFS 1998a
	1976-1978	714 (280-2,856)	Total	Taubert 1980; NMFS 1998a
	1976-1978	297 (267–618)	Total	Taubert 1980; NMFS 1998a
	1994	328 (188-1,264)	Adults	Kynard & Kieffer, unpubl.; NMFS unpubl.
	1994-2001	143 (14-360)	Spawning Adults	Kynard & Kieffer, unpubl.; NMFS unpubl.
Lower Connecticut River[e]	1988-1993	895 (799-1,018)	Adult	Savoy and Shake 1992; NMFS 1998a
Hudson River (NY)	1980	30,311	Total	Dovel 1979; NMFS 1998b
	1994-1997	61,057 (52,898-72,191)	Total	Bain *et al.* 2007
Delaware River (NJ, DE, PA)	1981-1984	12,796 (10,288-16,267)	Partial	Hastings *et al.* 1987
	1981-1984	14,080 (10,079-20,378)	Partial	Hastings *et al.* 1987
	1999-2003	12,047 (10,757-13,589)		Brundage and O'Herron 2003
Chesapeake Bay (MD, VA)				
Cape Fear River (NC)				
Winyah Bay (NC, SC)				
Santee River (SC)				
Cooper River (SC)	1996-1998	300	Adults	Cooke *et al.* 2004
ACE Basin (SC)				
Savannah River (SC, GA)		1,000 - 3,000	Adults	B Post, SCDNR 2003; NMFS unpubl.
Ogeechee River (GA)	1993	266 (236 – 300)		Weber 1996, 1998
	1993	361 (326 – 400)	Total	Rogers and Weber 1994, NMFS 1998b
	1999-2004	147 (104-249)		Fleming *et al.* 2003; NMFS unpubl.
Altamaha River (GA)	1988	2,862 (1,069 - 4,226)	Total	NMFS 1998a
	1990	798 (645 - 1,045)	Total	NMFS 1998a
	1993	468 (316 – 903)	Total	NMFS 1998a
		6,320 (4,387-9,249)	Total	DeVries 2006

Population (Location)[a]	Data Series	Abundance Estimate (C.I.)[b]	Population Segment	Reference
Satilla River (GA)				
Saint Mary's River (FL)				
Saint Johns River (FL)				FFWCC 2007c

[a] The original 19 populations identified by NMFS in the 1998 recovery plan are left aligned in this column. Estimates for a tributary or river segment are indented.

[b] Population estimates are established using different techniques and should be viewed with caution. In some cases, sampling biases may have violated the assumptions of the procedures used or resulted in inadequate representation of a population segment. Some estimates (e.g., those without confidence intervals or are depicted by ranges only) are the "best professional judgment" of researchers based on their sampling effort and success.

[c] Range represents total population estimates using four different techniques. All techniques suggest the population increased during the sampling period (see Savoy 2004 for more details).

[d] Above Holyoke Dam.

[e] Below Holyoke Dam.

Distribution. Shortnose sturgeon occur along the Atlantic Coast of North America, from the St. John River in Canada to the St. John's River in Florida. At the northern end of the species' distribution, the highest rate of gene flow (which suggests migration) occurs between the Ponobscot and Androscoggin Rivers. At the southern end of the species' distribution, populations south of the Pee Dee River appear to exchange between one and 10 individuals per generation, with the highest rates of exchange between the Ogeechee and Altamaha Rivers (Wirgin *et al.* 2005a). Wirgin *et al.* (2005) concluded that rivers separated by more than 250 miles were connected by very little migration while rivers separated by no more than 12 miles (such as the rivers flowing into coastal South Carolina) would experience high migration rates. Coincidentally, at the geographic center of the shortnose sturgeon range, there is a 250 mile stretch of river with no known populations occurring from the Delaware River, New Jersey to Cape Fear River, North Carolina (Kynard 1997a). However, shortnose sturgeon are known to occur in the Chesapeake Bay, and may be transients from the Delaware River via the Chesapeake (Skjeveland *et al.* 2000, Welsh *et al.* 2002) or remnants of a population in the Potomac River (Kynard *et al.* 2009).

Rogers and Weber (1995a), Kahnle *et al.* (1998a), and Collins *et al.* (2000b) concluded that shortnose sturgeon are extinct from the St. Johns River in Florida and the St. Marys River along the Florida and Georgia border. In 2002, a shortnose sturgeon was captured in the St. Johns River, Florida, suggesting either immigration or a small remnant population (FFWCC 2007c). Rogers and Weber (1995a) also concluded that shortnose sturgeon have become extinct in Georgia's Satilla River.

Habitat. Habitat use in fresh water during summer and winter months overlaps between adult and age-1 shortnose sturgeon (O'Herron II *et al.* 1993, Rogers and Weber 1995b, Kynard *et al.* 2000). Kynard *et al.* (2000) found that both age classes preferred deep-water curves with sand and cobble to higher velocity runs, particularly during winter months, and shifted to channel habitat as water temperatures rose in summer months. Many fish also exhibited diel movement patterns between deeper waters during the day and shallower waters at night (Kynard *et al.* 2000). During the summer, at the southern end of their range, shortnose sturgeon congregate in cool, deep, areas of rivers where adult and juvenile sturgeon can take refuge from high temperatures (Flournoy *et al.* 1992, Weber 1996a). In the Connecticut River and the Merrimack, Kynard *et al.* (2000) found shortnose generally used water about 3 meters deep, ranging from less than a meter to about 15 meters deep.

Movements. The general migratory strategy of shortnose sturgeon is similar to many fresh water and diadromous fishes, which probably optimizes feeding opportunities, minimizes losses due to unfavorable conditions (winter refuge migrations), and optimizes spawning success(Harden Jones 1968, Northcote 1978). Water temperatures, flow regimes, and barriers influence their movement patterns (Kynard 1997(Kynard *et al.* 2000). Adult shortnose sturgeon will migrate upstream to spawning areas in the spring or in the fall. Fish that migrate upstream in the fall generally overwinter in areas just downstream of spawning sites, while others including non-spawners will overwinter in estuarine waters. After spawning in the spring, spent (post-spawned) adults tend to migrate rapidly downstream to feeding areas in the estuary or to tidally influence fresh water during summer, when movement become limited, possibly due to stress asscoaited with high water temperatures and low oxygen concentrations (Dadswell *et al.* 1984a, Collins 2010).

In the Penobscot River, shortnose sturgeon occur year-round in estuarine waters, spending mid-October to mid-April in the upper estuarey and moving downstream to the middle estuary in spring and near the river mouth in summer (Fernandes *et al.* 2010). However, individuals move in and out of the Penobscot River, some traveling at least 150 km to other monitored rivers, such as the Kennebec (Fernandes *et al.* 2010).

Young-of-the year shortnose sturgeon move downstream after hatching, remaining in fresh water for about 1 year (Kynard 1997b). Initially, young shortnose sturgeon will reside short distances from spawning areas, and as they grow will tend to move further downstream (Dadswell *et al.* 1984a). By age 3 or older juvenile sturgeon will spend a large portion of their year at the salt- and fresh water interface of coastal rivers (NMFS 1998a).

Sturgeon are iteroparous, and based on limited data it appears that females sturgeon spawn every three to five years while males spawn every other year, although some may spawn in consecutive years (Kieffer and Kynard 1993, NMFS 1998a) (Dovel *et al.* 1992; Collins and Smith 1993). Spawning typically occurs during the spring, between mid-March and late May. Spawning areas are often located just below the fall line at the farthest accessible upstream reach of the river (NMFS 1998a). The onset of spawning may be cued to decreasing river discharge following the peak spring freshet, when water temperatures range from 8 to 12 °C and bottom water velocities range between 25-130 cm/s, although photoperiod appears to control spawning readiness (Dadswell *et al.* 1984a, NMFS 1998a).

Length at maturity is about 45-55 cm fork length for shortnose sturgeon and age at first spawning appears to vary along a latitudinal cline. According to spawning checks, it appears that male shortnose sturgeon in southern rivers will first spawn between ages 2 and 5, while fish as far north as the St. Johns River, Canada first spawn at about 10 to 11 years of age (Dadswell *et al.* 1984a, NMFS 1998a). Age at first spawning for female shortnose sturgeon varies from about age 6 to 18 years, like males, varying on a latitudinal cline (Dadswell *et al.* 1984a, NMFS 1998a). In general, fish in the northern portion of the species' range live longer than individuals in the southern portion of the species' range (Gilbert 1989b). The maximum age reported for a shortnose sturgeon in the St. John River in New Brunswick is 67 years (for a female), 40 years for the Kennebec River, 37 years for the Hudson River, 34 years in the Connecticut River, 20 years in the Pee Dee River, and 10 years in the Altamaha River (Gilbert 1989b). Male shortnose sturgeon appear to have shorter life spans than females (Gilbert 1989b).

Feeding. Like all sturgeon, shortnose have ventrally located, sucker-like mouths, structured for feeding on benthos. Foraging generally occurs in areas with abundant macrophytes, where juvenile and adult shortnose sturgeon feed on amphipods, polychaetes, and gasteropods (Dadswell *et al.* 1984b, Moser and Ross 1995, NMFS 1998a). Starting as larvae sturgeon use electroreception to identify prey. Olfaction and taste are also likely important to foraging, while vision is thought to play a minor role (Miller 2004). As adults, a significant portion of a shortnose sturgeon's diet may consist of freshwater mollusks (Dadswell *et al.* 1984b). Based on observations by Kynard *et al.* (2000), shortnose sturgeon will consume the entire mollusk, excreting the shell after ingestion.

Life span. Shortnose sturgeon in the northern portion of the species' range live longer than individuals in the southern portion of the species' range (Gilbert 1989a). The maximum age reported for a shortnose sturgeon in the St. John River in New Brunswick is 67 years (for a female), 40 years for the Kennebec River, 37 years for the Hudson River, 34 years in the Connecticut River, 20 years in the Pee Dee River, and 10 years in the Altamaha River (Dadswell *et al.* 1984b). Male shortnose sturgeon appear to have shorter life spans than females (Gilbert 1989a).

Status and trends. Shortnose sturgeon were listed as endangered on March 11, 1967, under the Endangered Species Preservation Act (32 FR 4001) and remained on the endangered species list with enactment of the ESA of 1973, as amended. Pollution and overfishing, including bycatch in the shad fishery, were listed as principal reasons for the species' decline. Shortnose sturgeon are listed as an endangered species throughout all of their range.

Northern shortnose sturgeon population abundances are generally larger than southern populations (Kynard 1997b). Updated population estimates also suggest that three of the largest populations (Kennebec, Hudson, and Delaware River) may be increasing or stable, although data is limited. The New York (Hudson River) shortnose sturgeon population is the largest extant population of this species and based on available data exhibits appears to have increased (NMFS 1998b, Bain *et al.* 2000). The most recent population estimate indicates this population consists of about 61,000-shortnose sturgeon (95% confidence interval [CI] was between 52,898 and 72,191 fish (Bain *et al.* 2000)). A comparison of the Bain estimate to the 1979/1980 population estimate of spawning adults by Dovel *et al.* (1992); about 13,000 fish) led Bain *et al.* (2000) to conclude that the population had made a dramatic increase (about 400 % increase) between 1979 and 1997. While still evidence of an increasing population, a comparison of total population estimates (30,000:60,000) would suggest the population has only doubled in size during the study years. Similarly, the Kennebec River population appears to be increasing. Early estimates suggest that the Kennebec River contained an estimated 7,200 adult shortnose sturgeon in 1977-81 (Squiers *et al.* 1982), while the most recent estimate for this population is about 9,500 fish (Squiers 2003), suggesting the population has increased by about 30 % in about a twenty year period.

Data from the Delaware River, suggests that the population may be stable. Brundage and O'Herron (2006) estimate that the current population for the Delaware River is 12,047 adult fish (1999-2003; 95% CI: 10,757-13,589), which is similar to the 1981/84 estimate by Hastings *et al.* (1987) of 12,796 fish (95% CI: 10,288-16367). The recent capture of several fish that were tagged as adults by Hastings *et al.* (1987) suggests that older fish may comprise a substantial portion of the Delaware River population. Despite their longevity, the viability of sturgeon populations is sensitive to variability in juvenile recruitment and survival (Anders *et al.* 2002, Gross *et al.* 2002, Secor *et al.* 2002). Although interannual variation in juvenile recruitment would be expected as a result of stochastic factors that influence spawning and egg/larval survival, if the mean population size does not change over the long-term it then it would appear there is sufficient juvenile survival to provide at least periodic recruitment into the adult age classes. Data on juvenile recruitment or age-1+ survival would, however, establish whether this population is at a stable equilibrium.

South of Chesapeake Bay, populations are relatively small compared to their northern counterparts. The largest of the southern populations of shortnose sturgeon is the Altamaha River population. Population estimates have been calculated several times for sturgeon in the Altamaha since 1993. Total population estimates shown pretty sizeable interannual variation is occurring; estimates have ranged from as low as 468 fish in 1993 to over 6,300 fish in 2006 (NMFS 1998a, DeVries 2006). The Ogeechee River is the next most studied river south of Chesapeake Bay, and abundance estimates indicate that the shortnose sturgeon population in this river is considerably smaller than that in the Altamaha River. The highest point estimate in 1993 using a modified Schnabel technique resulted in a total population estimate of 361 shortnose sturgeon (95% CI: 326-400). In contrast the most recent survey resulted in an estimate of 147 shortnose sturgeon (95% CI: 104-249), suggesting that the population may be declining.

Annual variation in population estimates in many basins is due to changes in yearly capture rates, which are strongly correlated with weather conditions (river flow and water temperatures). In "dry years" fish move into deep holes upriver of the saltwater/freshwater interface, which can make them more susceptible to gillnet sampling. Consequently, rivers with limited data sets among years and limited sampling periods within a year may not offer a realistic representation of the size or trend of the shortnose sturgeon population in the basin. As a whole, the data on shortnose sturgeon populations is rather limited and some of the differences observed between years may be an artifact of the models and assumptions used by the various studies. Long-term data sets and an open population model would likely provide for more accurate population estimates across the species range, and could provide the opportunity to more closely link strong-year classes to habitat conditions.

Throughout the specie's range there are other extant populations, or at least evidence that several other basins are used periodically. That is, shortnose sturgeon have been documented in the St. John's River (FL), the St. Mary's River, Chesapeake Bay, Potomac River, Piscataqua River, the Housatonic River, and others. Some basins probably previously contained shortnose populations, but recent sampling has been largely unsuccessful. Despite the occasional observations of shortnose sturgeon, populations may be extinct in several basins (e.g., St. John's (FL), St. Mary's, Potomac, Housatonic, and Neuse rivers). Those few fish that have been observed in these basins are generally presumed to be immigrants from neighboring basins. In some cases, (e.g. Chesapeake Bay) migratory information collected from tagged fish and genetic evidence confirms that fish captured in Chesapeake Bay were part of the Delaware River population (Grunwald *et al.* 2002, Wirgin *et al.* 2005b).

Natural threats. Yellow perch, sharks, and seals are predators of shortnose sturgeon juveniles (NMFS 1998a). The effects of disease and parasites are generally unknown.

Anthropogenic threats. Shortnose sturgeon have declined from the combined effects from the construction of hydropower and water diversion projects, dredging and blasting, water pollution, fisheries, and hatcheries. The construction of dams has resulted in substantial loss of shortnose sturgeon habitat along the Atlantic seaboard. In many cases dams divide shortnose sturgeon spawning habitat (e.g., Connecticut River, Penobscot River) and impede passage or block it completely. Where it has occurred, remediation measures, such as obstruction removal or modification to allow for fish passage have improved shortnose sturgeon habitat and likely improved productivity and more such modifications are planned in certain basins. For instance, with the breaching of the Bangor Dam in the Penobscot River in 1977 five river kilometers were opened to sturgeon and other anadromous fishes. With the recent signing of the Penobscot River Restoration Trust, access may be restored to another 29 km of habitat.

Historic fishery harvests, as well as the incidental harvest in current fisheries, have had lasting effects on shortnose sturgeon. In the late nineteenth and early twentieth centuries shortnose sturgeon commonly were harvested incidental to Atlantic sturgeon, the larger and more commercially valuable of these two sympatric sturgeon species (NMFS 1998a). The effects of these harvests may have latent and long-lasting impacts on some populations. At present there is no legal directed fishing effort for shortnose sturgeon in the United States, although some illegal poaching is suspected. Additionally, shortnose sturgeon are often caught incidental to other fisheries. For instance, shortnose are caught incidentally by bass anglers, and incidentally to alewife/gaspereau and shad fisheries in the St. John’s River in Canada, shad fisheries in the Altamaha River, Hudson River, and others (COSEWIC 2005, Bahn and Peterson 2009).

Habitat alterations from discharges, dredging or disposal of material into waterways, and other developmental activities along riverine and estuarine systems threaten shortnose sturgeon habitat. Periodic maintenance of harbors and rivers likely results in the direct take of some sturgeon, but perhaps of greater impact is the manner in which dredging alters benthic topography and community structure, and water quality (increase in suspended sediments). Shoreline development of liquefied natural gas facilities and alternative power sources also alters coastal habitats through changes in benthic communities by dredging, changes in water quality and water temperatures, and may increase the potential of ship strikes. In the Bay of Fundy, a tidal turbine killed at least three Atlantic salmon in the 1980s, and may be a threat to shortnose sturgeon as well (Dadswell and Rulifson 1994). Although currently the only example of this type of turbine in North America, increasing interests in finding alternative energy sources is expected to result in an increase in the number of marine turbines along the coast.

Fish kills have also been observed where estuaries are affected by urban and agricultural discharges that cause vegetative blooms and eutrophic conditions. Extreme declines in dissolved oxygen levels have occurred periodically throughout the species range. In the late 1960s and early 1970s, dissolved oxygen levels reached zero ppm in the Penobscot, Kennebec, and Androscoggin rivers and estuaries during the summer. Extreme low dissolved oxygen levels are have also plagued Chesapeake Bay. In most cases, dissolved oxygen levels have improved since through improved treatment and control of waste discharges in the past twenty years, but degraded conditions of benthos are still common in many estuaries throughout the species range as a result of this historic loading of organic materials, waste, and legacy toxins such as dioxin. As a result, shortnose sturgeon and other benthic organisms are regularly in direct contact with legacy pollutants, as well as a suite of common contaminants added from more current industrial and agricultural practices. Studies demonstrate that shortnose sturgeon carry a wide number of potentially hazardous contaminants. Individuals from the Delaware River contain numerous metals (mercury, aluminum, antimony, barium, cadmium, calcium, chromium, copper, iron, magnesium, manganese, nickel, potassium, sodium, vanadium, and zinc), PCDDs, PCDFs, PCBs, DDE, DDD, bis (2-ethylhexyl) phthalate, di-n-butylphthalate, and chlordane (ERC 2002). Most of these metals, PCDDs, PCDFs, and PCBs were also found in shortnose sturgeon in the Kennebec River (ERC 2003).

Climate change has the potential to affect sturgeon in similar, if not more significant, ways that it affects salmonids. Elevated air temperatures could lead to precipitation falling as rain instead of snow. Additionally, snow would likely melt sooner and more rapidly, potentially leading to greater flooding during melting and lower water levels at other times, as well as warmer river temperatures (ISAB 2007c). Although sturgeon can spawn over varied benthic habitat, they prefer localized depressions in riverbeds (Moyle *et al.* 1992a, Moyle *et al.* 1995a, Erickson *et al.* 2001b, Rien *et al.* 2001a). Increased extremes in river flow (i.e., periods of flooding and low flow) can alternatively disrupt and fill in spawning habitat that sturgeon rely upon (ISAB 2007c). If water flow is low during migration events, it is likely that new obstacles can impede or block sturgeon movement. As with other anadromous fishes, sturgeon are uniquely evolved to the environments that they live in. Because of this specificity, broad scale changes in environment can be difficult to adapt to, including changes in water temperature (Cech Jr. *et al.* 2000).

Sturgeon are also directly sensitive to elevated water temperatures. Temperature triggers spawning behavior. Warmer water temperatures can initial spawning earlier in a season for salmon and the same can be true for sturgeon (ISAB 2007c). If river and lake temperatures become anomalously warm, juvenile sturgeon may experience elevated mortality due to lack of cooler water refuges in freshwater habitats. If temperature rise beyond thermal limits for extended periods, habitat can be lost; this could be the case if southern habitats warm, resulting in range loss(Lassalle *et al.* 2010). Apart from direct changes to sturgeon survival, altered water temperatures may disrupt habitat, including the availability of prey (ISAB 2007c). Warmer temperatures may also have the effect of increasing water use in agriculture, both for existing fields and the establishment of new ones in once unprofitable areas (ISAB 2007c). This means that streams, rivers, and lakes will experience additional withdrawal of water for irrigation and increasing contaminant loads from returning effluent. Overall, it is likely that global warming will increase pressures on sturgeon survival and recovery.

Critical habitat. Critical habitat has not been established for shortnose sturgeon.

Smalltooth sawfish

Description of the species. The smalltooth sawfish is a tropical marine and estuarine elasmobranch fish (sharks and rays) that has been reported to have a circumtropical distribution. Although they are rays, sawfish physically more resemble sharks, with only the trunk and especially the head ventrally flattened. Smalltooth sawfish are characterized by their "saw," a long, narrow, flattened rostral blade with a series of transverse teeth along either edge.

Distribution. In the western Atlantic, the smalltooth sawfish has been reported from Brazil through the Caribbean and Central America, the Gulf of Mexico, and the Atlantic coast of the US. The smalltooth sawfish has also been recorded from Bermuda (Bigelow and Schroeder 1953). Forms of smalltooth sawfish have been reported from the eastern Atlantic in Europe and West Africa; the Mediterranean; South Africa; and the Indo-West Pacific, including the Red Sea, India, Burma, and the Philippines (Bigelow and Schroeder 1953, Van der Elst 1981, Compagno and Cook 1995). Whether populations outside of the Atlantic are truly smalltooth sawfish or closely related species is unknown (Adams and Wilson 1995). Pacific coast records of smalltooth sawfish off Central America need confirmation (Bigelow and Schroeder 1953, Compagno and Cook 1995).

The range of the smalltooth sawfish in the Atlantic has contracted markedly over the past century. The northwestern terminus of their Atlantic range is located in the waters of the eastern US. Historic capture records within the US range from Texas to New York. Water temperatures no lower than 61°F to 64.4°F and the availability of appropriate coastal habitat serve as the major environmental constraints limiting the northern movements of smalltooth sawfish in the western North Atlantic (Simpfendorfer 2001). As a result, most records of this species from areas north of Florida occur during spring and summer periods (May to August) when inshore waters reach appropriately high temperatures. The data also suggest that smalltooth sawfish may utilize warm water outflows of power stations as thermal refuges during colder months to enhance their survival or become trapped by surrounding cold water from which they would normally migrate. Almost all occurrences of smalltooth sawfish in warm-water outflows were during the coldest part of the year, when water temperatures in these outfalls are typically well above ambient temperatures.

Movement. Historic records of smalltooth sawfish indicate that some large mature individuals migrated north along the US Atlantic coast as temperatures warmed in the summer and then south as temperatures cooled (Bigelow and Schroeder 1953). Recent Florida encounter data, however, do not suggest such migration. Only two smalltooth sawfish have been recorded north of Florida since 1963 (the first was captured off of North Carolina in 1999 and the other off Georgia 2002) but it is unknown whether these individuals resided in Georgia and North Carolina waters annually or if they had migrated north from Florida (Schwartz 2003b, Burgess unpublished data). Given the very limited number of encounter reports from the east coast of Florida, Simpfendorfer and Wiley (2004) hypothesize the population previously undertaking the summer migration has declined to a point where the migration is undetectable or does not occur.

Most specimens captured along the Atlantic coast north of Florida have also been large (>2.7 m) adults and likely represent seasonal migrators, wanderers, or colonizers from a core population(s) to the south rather than being members of a continuous, even-density population (Bigelow and Schroeder 1953). It is likely that these individuals migrated southward toward Florida as water temperatures declined in the fall, as there is only one winter record from the Atlantic coast north of Florida. Based on smalltooth sawfish encounter data, the current core range for the smalltooth sawfish is from the Caloosahatchee River, Florida, to Florida Bay (NMFS 2000, Simpfendorfer and Wiley 2004).[3]

Habitat. Smalltooth sawfish are euryhaline, occurring in waters with a broad range of salinities from freshwater to full seawater (Simpfendorfer 2001). Younger, smaller individuals tend to inhabit very shallow mud banks and tides are a major factor in their movement (Simpfendorfer *et al.* 2010). At this size smalltooth sawfish spend the vast majority of their time on shallow mud or sand banks that are less than 1 foot (30 cm) deep. As they grow, juveniles tend to occupy deeper habitat, but shallow areas (<1 m depth) remain preferred habitat; juveniles also expand their ranges, whereas small individuals have very restricted ranges (Simpfendorfer *et al.* 2010). Acoustic tracking studies have shown that at this size sawfish will remain associated with the same mud bank over periods of several days. These banks are often very small and daily home range sizes can be of the magnitude of 100–1,000 m^2 (Simpfendorfer 2003b). Acoustic monitoring studies have shown that juveniles have high levels of site fidelity for specific nursery areas for periods up to almost 3 months (Wiley and Simpfendorfer 2007). Their occurrence in freshwater is suspected to be only in estuarine areas temporarily freshwater from receiving high levels of freshwater input. Many encounters are reported at the mouths of rivers or other sources of freshwater inflows, suggesting estuarine areas may be an important factor in the species distribution (Simpfendorfer and Wiley 2004).

Information on juvenile smalltooth sawfish indicates that they prefer shallow euryhaline habitats adjacent to red mangroves (NMFS 2006h). They do still have a preference for shallow water, remaining in depths mostly less than 90 cm. Several sawfish approximately 150 cm in length fitted with acoustic tags have been relocated in the same general areas over periods of several months, suggesting a high level of site fidelity (Simpfendorfer 2003). The daily home ranges of these animals are considerably larger (1–5 km^2) than for the very small sawfish and there is less overlap in home ranges between days.

Smalltooth sawfish are most common in shallow coastal waters less than 25 m (Bigelow and Schroeder 1953, Adams and Wilson 1995). Indeed, the distribution of the smallest size classes of smalltooth sawfish indicate that nursery areas occur throughout Florida in areas of shallow water, close to shore and typically associated with mangroves (Simpfendorfer and Wiley 2004). However, encounter data indicate there is a tendency for smalltooth sawfish to move offshore and into deeper water as they grow. Larger animals are more likely to be found in deeper waters. Poulakis and Seitz (2004b) reported that almost all of the sawfish <3 m in length were found in water less than 10 m deep and 46% of encounters individuals >3 m in Florida Bay and the Florida Keys were reported at depths between 70 to 122 m. Since large animals are also observed in very shallow waters, it is believed that smaller (younger) animals are restricted to shallow waters, while large animals roam over a much larger depth range (Simpfendorfer 2001). Recent data from sawfish encounter reports and from satellite tagging indicate mature animals

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[3] See the 2006 Draft Recovery Plan for more detailed information on the historic and current distribution of smalltooth sawfish in four regions of the eastern U.S. This information is based on the Status Review Team's analysis and the more recent encounter database research {Poulakis, 2004 #1266;Seitz, 2002 #1426;Simpfendorfer, 2004 #1465;Simpfendorfer, 2004 #1465}.

occur regularly in waters in excess of 164 feet (Poulakis and Seitz 2004a, Simpfendorfer and Wiley 2004).

Growth and reproduction. As in all elasmobranchs, fertilization is internal. Bigelow and Schroeder (1953) report the litter size as 15 to 20. Simpfendorfer and Wiley (2004), however, caution this may be an overestimate, with recent anecdotal information suggesting smaller litter sizes (about ten). Smalltooth sawfish mating and pupping seasons, gestation, and reproductive periodicity are all unknown. Gestation and reproductive periodicity, however, may be inferred based on that of the largetooth sawfish, sharing the same genus and having similarities in size and habitat. Thorson (1976) reported the gestation period for largetooth sawfish was approximately five months and concluded that females probably produce litters every second year.

Bigelow and Schroeder (1953) describe smalltooth sawfish as generally about 61 cm long at birth and growing to a length of 5.5 m or greater. Recent data from smalltooth sawfish caught off Florida, however, demonstrate young are born at 76 to 87 cm (Simpfendorfer and Wiley 2004), with males reaching maturity at approximately 2.7 m and females at approximately 3.6 m (Simpfendorfer 2002). A recent study by Simpfendorfer suggests rapid juvenile growth occurs during the first two years after birth (Simpfendorfer 2008). First year growth is 65-85 cm and second year growth is 48-68 cm. Growth rates beyond two years are uncertain; however, the average growth rate of captive smalltooth sawfish has been reported between 13.9 and 19.6 cm per year. The maximum reported size of a smalltooth sawfish is 24.9 feet (Last and Stevens 1994), but the maximum size normally observed is 7.6 m (Adams and Wilson 1995). No formal studies on the age and growth of the smalltooth sawfish have been conducted to date, but growth studies of largetooth sawfish suggest slow growth, late maturity (10 years) and long lifespan (25 to 30 years Thorson 1982, Simpfendorfer 2000b). These characteristics suggest a very low intrinsic rate of increase (Simpfendorfer 2000b).

Simpfendorfer estimated intrinsic rates of natural population increase at 0.08 to 0.13 per year and population doubling times from 5.4 to 8.5 years (Simpfendorfer 2000a).

Feeding. Smalltooth sawfish feed primarily on fish, with mullet, jacks, and ladyfish believed to be their primary food resources (Simpfendorfer 2001). In addition to fish, smalltooth sawfish also prey on crustaceans (mostly shrimp and crabs), which are located by disturbing bottom sediment with their saw (Norman and Fraser 1937, Bigelow and Schroeder 1953).

Status and trends. The US smalltooth sawfish distinct population segment (DPS) was listed as endangered under the ESA on April 1, 2003 (68 FR 15674). The smalltooth sawfish is the first marine fish to be listed in the US. Despite being widely recognized as common throughout their historic range up until the middle of the 20th century, the smalltooth sawfish population declined dramatically during the middle and later parts of the century. The decline in the population of smalltooth sawfish is attributed to fishing (both commercial and recreational), habitat modification, and sawfish life history. Large numbers of smalltooth sawfish were caught as bycatch in the early part of this century. Smalltooth sawfish were historically caught as bycatch in various fishing gears throughout their historic range, including gillnet, otter trawl, trammel net, seine, and to a lesser degree, handline. Frequent accounts in earlier literature document smalltooth sawfish being entangled in fishing nets from areas where smalltooth sawfish were once common but are now rare (Evermann and Bean 1898). Loss and/or degradation of habitat contributed to the decline of many marine species and continue to impact the distribution and abundance of smalltooth sawfish. Simpfendorfer (2001) estimated that the US population size is currently less than 5% of its size at the time of European settlement. One dataset from shrimp trawlers off Louisiana from the late 1940s through the 1970s suggests a rapid decline in the species from the period 1950-1964 (NMFS 2006h).

Seitz and Poulakis (2002) and Poulakis and Seitz (2004a) documented recent (1990 to 2002) occurrences of sawfish along the southwest coast of Florida, and in Florida Bay and the Florida Keys, respectively and includes a total of 2,969 smalltooth sawfish encounters. Mote Marine Laboratory also maintains a smalltooth sawfish public encounter database, established in 2000 to compile information on the distribution and abundance of sawfish. A total of 434 sawfish encounters have been validated since 1998, most from recreational fishers (Simpfendorfer and Wiley 2004). Dr. Simpfendorfer reluctantly gives an estimate of 2,000 individuals based on his four years of field experience and data collected from the public, but cautions that actual numbers may be plus or minus at least 50%.

The majority of smalltooth sawfish encounters today are from the southwest coast of Florida between the Caloosahatchee River and Florida Bay. Outside of this core area, the smalltooth sawfish appears more common on the west coast of Florida and in the Florida Keys than on the east coast, and occurrences decrease the greater the distance from the core area (Simpfendorfer and Wiley 2004). The capture of a smalltooth sawfish off Georgia in 2002 is the first record north of Florida since 1963. New reports during 2004 extend the current range of the species to Panama City, offshore Louisiana (south of Timbalier Island in 100 feet of water), southern Texas (unconfirmed), and the northern coast of Cuba.

The abundance of juveniles encountered, including very small individuals, suggests that the population remains reproductively active and viable (Seitz and Poulakis 2002, Simpfendorfer 2003a, Simpfendorfer and Wiley 2004). The declining numbers of individuals with increasing size is consistent with the historic size composition data (G. Burgess, pers. comm. in Simpfendorfer and Wiley 2004). This information and recent encounters in new areas beyond the core abundance area suggest that the population may be increasing. From 1989-2004, smalltooth sawfish relative abundance has increased by about 5 percent per year (Carlson *et al.* 2007). However, recovery of the species expected to be slow on the basis of the species' life history and other threats to the species remaining (see below), the population's future remains tenuous. Based on genetic sampling, the estimates of current effective population size are between 269.6 and 504.9 individuals (95% CI 139.3 – 1,515). (NMFS 2011l). This number is usually 25-50% of census population size (breeding adults) in elasmobranchs, so it is likely that the breeding population consists of high hundreds to low thousands of individuals (NMFS 2011l).

Natural threats. The primary natural threat to smalltooth sawfish survival is the species low reproductive rate. In the face of reduced population sizes, this biological parameter means that recovery, at best, will be slow, and that catastrophic perturbations can have severer consequences to recovery.

Anthropogenic threats. Smalltooth sawfish decline has been largely due to fisheries interaction (see NMFS 2006g for a review). The distinctive “saw” can easily become entangled in a variety of commercial and recreational fishing gear, resulting in drowning or injury. Even when individuals that have been entangled are retrieved alive, individuals may be killed for curio collection of the saw, fear of injury from fisherman, or injured from the gear or handling during gear removal. However, additional anthropogenic impacts result from habitat loss. Destruction of mangrove habitat, dredging, trawling and filling, and loss of reef habitat have negative impacts on all life stages of smalltooth sawfish. Although a concern, pollution impacts on particularly reproductive biology are unknown. However, habitat degradation due to runoff containing pesticides, eutrophying agents, and other contaminants can also have a negative impact on smalltooth sawfish habitat.

Critical habitat. On September 2, 2009, critical habitat was designated for smalltooth sawfish along the central and southwest coast of Florida (74 FR 45353). Although PCEs were not identified, the mangrove and adjacent shallow euryhaline habitat are important nursery habitat for smalltooth sawfish.

Bocaccio

Description of the species. The bocaccio is a rockfish species that genetic analyses suggest is composed of two distinct populations (Wishard *et al.* 1980, Matala *et al.* 2004). 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 (Hartmann 1987, Love *et al.* 2002, Emery *et al.* 2006). 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.* 1983a, Love *et al.* 1991). After a few weeks, fish move into slightly deeper waters of 18-30 m and occupy rocky reefs (Feder *et al.* 1974, Carr 1983, Eschmeyer *et al.* 1983a, 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.* 2002, Love *et al.* 2005, Love and York 2005, Love *et al.* 2006). 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 (Hart 1973b, Echeverria 1987, 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 (Hart 1973b, Echeverria 1987, Love *et al.* 2002, MacCall 2002b). Mating occurs between August and November, with larvae born between January and April (Lyubimova 1965, Moser 1967, Westrheim 1975, Wyllie Echeverria 1987, Love *et al.* 2002, MacCall and He 2002b).

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 (Moser 1967, Matarese *et al.* 1989, Woodbury and Ralston 1991, Love *et al.* 2002, MacCall and He 2002b, MacCall 2003). 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 (Eschmeyer *et al.* 1983a, Halstead *et al.* 1990, Ralston and Ianelli 1998, Love *et al.* 2002, Andrews *et al.* 2005, Piner *et al.* 2006). 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 (Reilly *et al.* 1992, Love *et al.* 2002).

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.* 2009).

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 (Sakuma and Ralston 1995, MacCall 1996, Love *et al.* 1998b, Moser *et al.* 2000b). 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 and He 2002a, MacCall 2008, 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 known predators of bocaccio and other rockfish species (Love *et al.* 2002, Beaudreau and Essington 2007, Lance and Jeffries 2007). Bocaccio and other rockfish appear to be negatively influenced by El Niño conditions, possibly reducing available prey supply (Moser *et al.* 2000a, Harvey 2005).

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 (WDF 1975-1986, Palsson *et al.* 2008). 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 US 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.* 1983a, Richards *et al.* 1985, Love *et al.* 1991). 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.* 1983a, 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, Richards 1986, Love *et al.* 1991, O'Connell and Carlisle 1993, 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 (Hitz 1962, DeLacy *et al.* 1964, Westrheim 1975, O'Connell 1987, Wyllie Echeverria 1987, Lea *et al.* 1999). 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)(MacGregor 1970, Hart 1973b). 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, Matarese *et al.* 1989, Moser 1996a, Love *et al.* 2002). 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, Westrheim and Harling 1975, Rosenthal *et al.* 1982, Love *et al.* 2005, 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 (Love 1978, Coombs 1979, DeMott 1983).

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 (Love *et al.* 2002, Beaudreau and Essington 2007, Lance and Jeffries 2007). Yelloweye and other rockfish appear to be negatively influenced by El Niño conditions, possibly reducing available prey supply (Moser *et al.* 2000a, Harvey 2005, Black 2009). 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 (Yamanaka and Kronlund 1997, Wallace 2007).

Anthropogenic threats. Overfishing is considered the primary cause of yelloweye rockfish decline throughout their range, including in Washington State and British Columbian waters (Wallace 2007, NMFS 2008d). 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 (Miller and Lea 1972, Hart 1973b, Cailliet *et al.* 2000, Love *et al.* 2002).

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 (Miller and Geibel 1973, Love *et al.* 1991, Cailliet *et al.* 2000, Love *et al.* 2002). 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 (Phillips 1960, Boehlert 1980, Lamb and Edgell 1986, Rosenthal *et al.* 1998, Starr 1998, Cailliet *et al.* 2000, Johnson *et al.* 2003, Methot and Stewart 2005, 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 (Moser 1996b, Methot and Stewart 2005, 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 (Vetter and Lynn 1997, Methot and Stewart 2005). 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 (Gascon and Miller 1981, DeMott 1983, Casillas *et al.* 1998, 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)(Hart 1973b, Westrheim and Harling 1975, Wyllie Echeverria 1987, Barss 1989).

Growth and development. When born, larvae are 3.6-4.0 mm in length and take from 1-4 months to develop into juveniles (Waldron 1968, Richardson and Laroche 1979, Stahl-Johnson 1985, Moser 1996a, Krigsman 2000, Love *et al.* 2002). 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 (Westrheim and Harling 1975, Boehlert and Kappenman 1980, Lenarz and Echeverria 1991, STAT 1999). 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 (Cailliet *et al.* 2000, Cailliet *et al.* 2001, Andrews *et al.* 2007). Maximum reported sizes are 76 cm and 4.5 kg (Boehlert 1980, IGFA

1991, Williams *et al.* 1999, Love *et al.* 2002, Methot and Stewart 2005).

Foraging. Canary rockfish prey upon different species as they age. Larvae are planktivores, consuming invertebrate eggs, copepods, and nauplii (Moser and Boehlert 1991, Love *et al.* 2002). 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 (Merkel 1957, Miller and Geibel 1973, Morejohn *et al.* 1978, Roberts 1979, Antonelis Jr. and Fiscus 1980, Ainley *et al.* 1981, Rosenthal *et al.* 1982, Stevens and Miller 1983, Love *et al.* 1991, Beaudreau and Essington 2007, Lance and Jeffries 2007). Canary and other rockfishes appear to be negatively influenced by El Niño conditions, possibly reducing available prey supply (Moser *et al.* 2000a, Harvey 2005).

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.

Pacific eulachon

Description of the Species. Eulachon that spawn in rivers south of the Nass River of British Columbia to the Mad River of California have been separated into the proposed listing of southern DPS eulachon. This is based upon timing of runs related to temperature, genetic distinctions, size at maturity, and ecological features of both oceanic and freshwater environments. Differences may also exist in mean number of vertebrae (Hart and McHugh 1944, McLean *et al.* 1999, Hay and McCarter 2000, McLean and Taylor 2001, Beacham *et al.* 2005). Like salmon, eulachon likely imprint upon chemical cues in their natal system. However, unlike salmon, hatchlings spend far less time in freshwater systems and likely retain homing only to the estuarine system that their natal river drains to. Based upon this, the smallest stock unit is likely the estuary that natal streams drain (Hay and McCarter 2000, Beacham *et al.* 2005). Specific spawning rivers within the natal system are likely selected based upon environmental conditions at that time of return (Hay and Beacham 2005).

Distribution. Eulachon, or candlefish, are small smelt native to eastern North Pacific waters from the Bering Sea to Monterey Bay, California, or from 61º N to 31º N (Hart and McHugh 1944, Eschmeyer *et al.* 1983b, Minckley *et al.* 1986, Hay and McCarter 2000). However, the southern extent of their distribution has receded northward over the past several decades. The lower Columbia River mainstem provides spawning and incubation sites, and a large migratory corridor to spawning areas in the tributaries. Prior to the construction of Bonneville Dam, eulachon ascended the Columbia River as far as Hood River, Oregon. Major tributaries that support spawning runs include the Grays, Skamokawa, Elochoman, Kalama, Lewis and Sandy rivers.

Growth and reproduction. Although primarily marine, eulachon return to freshwater to spawn. Adult eulachon have been observed in several rivers along the west coast, including the Umpqua and Rogue rivers in Oregon, California's Humboldt Bay, Klamath, Mad, Russian, and Sacramento rivers as well as Redwood Creek, and Washington's Puget Sound, Hood Canal, Bear, Naselle, Nemah, Wynoochee, Quinault, Queets, and Nooksack rivers (Odemar 1964, Moyle 1976b, Minckley *et al.* 1986, Emmett *et al.* 1991b, Jennings 1996, Wright 1999, Larson and Belchik 2000, Musick *et al.* 2000a, WDFW and ODFW 2001). Spawning has been documented in the Elwha River on the Strait of Juan de Fuca, but sightings or spawning in these Oregonian and Washingtonian rivers is very limited or unknown (Wright 1999, Shaffer *et al.* 2007). For southern DPS eulachon, most spawning is believed to occur in the Columbia River and its tributaries (Grays, Skamokawa, Elochoman, Kalama, Lewis, and Sandy rivers), with secondary levels in the Mad and Klamath rivers, as well as sporadic production in other Oregonian and Washingtonian rivers (Emmett *et al.* 1991b, Musick *et al.* 2000a, WDFW and

ODFW 2001). Southern DPS fish likely take less time to mature than do fish from more northerly rivers and generally spawn earlier in southern portions of their range than in northern rivers (Clarke *et al.* 2007). Most eulachon die following migration and spawning, but some have outmigrated and returned for a second spawning event.

North of Washington State, eulachon spawning is supported in two major river systems: the Fraser and Yukon (Hart and McHugh 1944, Richardson *et al.* 2000, NPS 2008). Although 33 rivers have been documented to support spawning in British Columbia alone, only half of these are considered to have sustained spawning (Hay and McCarter 2000, Willson *et al.* 2006).

Spawning takes place at differential time and temperatures, depending upon the river system involved (Willson *et al.* 2006). In the Columbia River and further south, spawning occurs from late January to March, although river entry occurs as early as December (Hay and McCarter 2000). The peak of eulachon runs in Washington State is from February through March. Fraser River spawning is significantly later, in April and May (Hay and McCarter 2000). Rivers of northern British Columbia host eulachon runs in late February through March (the same time as Washington State peak runs) and peaks from mid-March to mid-May (Wydoski and Whitney 1979a). Alaskan runs occur in May and river entry may extend into June (Hay and McCarter 2000). Although a trend of earlier spawning in southerly rivers is apparent, euchalon spawning may occur as early as January in Alaska and as late as May in northern California rivers. Additionally, smaller-scale resolution of British Columbia spawning supports earlier runs (February to early March) in northern territorial rivers (i.e., Nass River) and later spawning (April and May) in more southerly rivers (i.e., Fraser River).

The timing of euchalon entry into spawning rivers is likely tied to water temperature and tidal cycles (Ricker *et al.* 1954, Bishop *et al.* 1989, PRFR 1998, WDFW and ODFW 2001, Lewis *et al.* 2002, Spangler 2002). Water velocity may be a factor in spawning timing, where water velocity greater than 0.9 miles per hour can limit upstream migration (Lewis *et al.* 2002). Spawning normally occurs when water temperature is between 39º and 50º F, although Stikine and Nass River spawning is known to occur at cooler temperatures (Smith and Saalfeld 1955, Langer *et al.* 1977, Franzel and Nelson 1981, WDFW and ODFW 2001). Similarly, deviations in spawning while temperatures are warmer have been noted for the Susitna River in Alaska (Barrett *et al.* 1984, Vincent-Lang and Queral 1984). Adults are known to be sensitive to temperature changes; in the Cowlitz River, an increase in temperature from 41º to 52º F over several days was followed by a 50% mortality in adults and failure to spawn (Blahm and McConnell 1971).

Adults may migrate up to 100 miles upstream to reach spawning grounds (Hart and McHugh 1944). Males tend to arrive on spawning grounds earlier than females and tend to stay longer, making them more susceptible to commercial and recreational fisheries (Hart and McHugh 1944). However, males outnumber females by a roughly 2:1 margin. Eulachon sperm is viable for only minutes and a key factor of eulachon spawning may be male grouping *en mass* to broadcast their sperm. Once milt reaches downstream females, they release 17,000 to 60,000 eggs (generally 25,000 on average) at which time fertilization occurs. Females lay eggs over sand, course gravel, or detritial substrate. However, like abalone, this method requires high eulachon density to ensure fertilization. Density may also impact the ability of eulachon to undergo synchronization of spawning, mate choice, and gonadal sterol levels. Eggs attach to gravel or sand and incubate for 30 to 40 days after which larvae drift to estuaries and coastal marine waters (Wydoski and Whitney 1979a). Larvae develop into juveniles and, after being in estuaries and marine waters for 3 to 5 years, migrate back to natal basins to spawn.

Eulachon generally die following spawning (Scott and Crossman 1973b). Maximum know lifespan is 9 years of age, but 20 to 30% of individuals live to 4 years and most individuals survive to 3 years of age, although spawning has been noted as early as 2 years of age (Barraclough 1964, Parente and Snyder 1970, Langer *et al.* 1977, Wydoski and Whitney 1979a, Barrett *et al.* 1984, Hugg 1996, Hay and McCarter 2000, WDFW and ODFW 2001). However, the age distribution of spawners varies between river and from year-to-year (Willson *et al.* 2006).

Habitat. Adult eulachon are found in coastal and offshore marine habitats possibly to 2,000 feet deep, but more frequently between 50 and 600 feet deep (Allen and Smith 1988, Hay and McCarter 2000, Willson *et al.* 2006). Following hatching in freshwater, larvae and juveniles become thoroughly mixed in coastal waters generally less than 50 feet deep, but can be found as far down as 600 feet deep (Barraclough 1964, Hay and McCarter 2000).

Foraging. Larval and post larval eulachon prey upon phytoplankton, copepods, copepod eggs, mysids, barnacle larvae, worm larvae, and other eulachon larvae until they reach adult size (WDFW and ODFW 2001). At this time, the primary prey of eulachon are obtained by filer feeding on copepods and euphausiids, including *Thysanoessa* spp., unidentified malacostracans, and cumaceans (Smith and Saalfeld 1955, Barraclough 1964, Wydoski and Whitney 1979a, Drake and Wilson 1991, Sturdevant *et al.* 1999, Hay and McCarter 2000).

Status and trends. The southern DPS of eulachon was proposed for listing as a threatened on March 13, 2009 (74 FR 10857). It is considered to be at moderate risk of extinction throughout the DPSs range. This is likely due to a variety of factors, including predation, commercial and recreational fishing pressure (directed and bycatch), and loss of habitat. Population decline is anticipated to continue as a result of climate change and bycatch in commercial shrimp fisheries. However, as highly fecund fish, eulachon have the ability to rebound quickly if given the opportunity, a feature that is likely necessary to withstand significant predation pressure and high mortality likely experienced by pelagic larvae (Bailey and Houde 1989). The median minimum population doubling time for eulachon is estimated to be 1.4 to 4.4 years (Musick *et al.* 2000a).

Eulachon formerly experienced widespread, abundant runs and have been a staple of native American diets for centuries along the northwest coast. However, such runs that were formerly present in several California rivers as late as the 1960s and 1970s (i.e., Klamath River, Mad River, and Redwood Creek) no longer occur (Larson and Belchik 2000). This decline likely began in the 1970s and continued until, in 1988 and 1989, the last reported sizeable run occurred in the Klamath River and no fish were found in 1996, although a moderate run was noted in 1999 (Larson and Belchik 2000, Moyle 2002b). Eulachon have not been identified in the Mad River and Redwood Creek since the mid-1990s, although effort here may be low or non-existent (Moyle 2002b). Despite a brief period of improved returns in 2001–2003, the returns and associated commercial landings have again declined to the very low levels observed in the mid-

1990s (JCRMS 2009) , and since 2005, the fishery has operated at the most conservative level allowed in the management plan (JCRMS 2009).

As mentioned, the Columbia River and its tributaries form the backbone of southern DPS spawning grounds. Historically, this system likely supported half of all spawning abundance, but has declined precipitously since the early and mid 1990s (ODFW and WDFW 2007). Although regulations on commercial and recreational catches have been implemented throughout southern DPS freshwater range, commercial catch records suggest populations are a small fraction of their former abundance (998 metric tons from 1936 to 1992 on average versus 91 metric tons annually from 1993 to present; (ODFW and WDFW 2007)). The Fraser River also represents a key spawning area for the southern DPS of eulachon and also experienced extensive declines in abundance to a presently historic low (10 metric tons of commercial catch, or roughly 300,000 individuals)(Schweigert *et al.* 2007, DFO 2008). This is a decline of over 90% since the early 2000s, or roughly two fish generations.

Natural threats. Eulachon are heavily predated upon by numerous marine and terrestrial species. These include: fishes (green, white, and mammoth sturgeon, salmon, piked and spiny dogfish, sablefish, salmon sharks, lingcod, arrowtooth flounder, Dolly Varden, Pacific halibut, Pacific hake, and Pacific cod), marine mammals (baleen whales, porpoises, orcas, dolphins, fur seals, harbor seals, and Steller sea lions), birds (harlequin ducks, pigeon guillemots, common murres, mergansers, cormorants, gulls, and eagles), and terrestrial mammals (Clemens *et al.* 1936, Hart 1973a, Scott and Crossman 1973b, Jeffries 1984, Drake and Wilson 1991, Yang and Nelson 1999, Willson *et al.* 2006). The high fat content of eulachon make them a particularly valuable prey for white sturgeon in the Columbia and Fraser rivers during winter (Willson *et al.* 2006).

Anthropogenic threats. Eulachon face numerous anthropogenic effects that have likely lead to their decline. Fisheries harvests are likely a major contributor to eulachon decline. The best available information for catches comes from the Columbia River, where catches have been as high as 5.7 million pounds per year, but averaged near 2 million pounds from 1938 to 1993 (Wydoski and Whitney 1979a). Since 1993, catches have not exceeded 1 million pounds annually and the median catch has been 43,000 pounds (97.7% reduction in catch), even when effort is accounted for (WDFW and ODFW 2001). Efforts to quantify fish that will be returning to the Columbia River to spawn in subsequent years have suggested significant variation in run size (100 metric tons to greater than 4,000 metric tons), but average less than 1,000 metric tons. Catch from sport fisheries is also high (Wydoski and Whitney 1979a). Outside of the Columbia River, the next highest landing size has been in the Fraser River, but catch here is nearly ten times smaller than in the Columbia (Hay *et al.* 2003). Bycatch from shrimp trawling along US and Canadian coasts has also been high, composing up to 28% of the total catch by weight, triggering periodic closures of the fishery in some years (Hay and McCarter 2000, DFO 2008).

Perhaps the largest threat to eulachon survival and recovery is changing ocean conditions associated with global warming. Increases in ocean temperatures have already occurred and will likely to continue to impact eulachon and their habitats. For example, changes in glacial melt throughout the Alaskan, British Columbian, Washingtonian, and Oregonian coasts will likely change flow patterns in major rivers, such as the Fraser, that are important for eulachon spawning (Morrison *et al.* 2002). Alterations in the hydrology of the Columbia River include decreased snowpack, increased peak flows, decreased base flow, and increased water temperatures, which will pose similar challenges to eulachon spawning as in the Fraser River (ISAB 2007b). Reductions in glacier sizes may also shift peak flow throughout regional rivers (Meier *et al.* 2003, Mote 2003, Barry 2005). In the marine environment, eulachon rely upon cool or cold ocean regions and the invertebrate communities therein (Willson *et al.* 2006). As with El Niño and La Niña events, warming ocean temperatures will likely alter these communities, making it more difficult for eulachon and their larvae to locate or capture prey (Roemmich and McGowan 1995, Zamon and Welch 2005a). Warmer waters could also allow for the northward expansion of eulachon predator and competitor ranges, increasing an already high predation pressure on the species (Rexstad and Pikitch 1986, McFarlane *et al.* 2000, Phillips *et al.* 2007).

Construction projects have also had a negative impact on eulachon stocks. Dams, such as the Bonneville Dam on the Columbia River, have blocked eulachon from moving into former spawning habitat (Smith and Saalfeld 1955). Such damming projects also alter sedimentation and flow dynamics that eulachon have developed around in their evolution. River substrate composition, likely critical to successful spawning, is also altered by dams. The impoundment of water tends to raise water temperatures; a factor that spawning eulachon are particularly sensitive to (NMFS 2008c). Sediment retention structures constructed in response to the Mount St. Helens eruption to limit sediment transport downstream that could block navigation, have been correlated with reduced eulachon runs in subsequent years (Lou Reebs, pers. comm. *in* 74 FR 10857). Dredging activities likely destroy eggs and remove the benthic substrates they rely upon. Although poorly known, eulachon ecotoxicological studies support contaminant burdens, particularly of arsenic and lead, which can be high (Futer and Nassichuk 1983, Rogers *et al.* 1990, EPA 2002).

Critical habitat. On October 20, 2011, the NMFS proposed to designate critical habitat for the southern DPS of eulachon, including roughly 539 km of riverine and estuarine habitat in Washington State (Grays, Skamokawa, Elochoman, Cowlitz, Kalama, Toutle, Lewis, Quinault, and Elwa rivers/creeks), Oregon (Columbia River), and California (Mad, Klamath, Redwood, Umpqua, and Sandy rivers as well as Tenmile Creek)(76 FR 65324). These areas contain physicalor biological features essential to the conservation of the DPS, including (1) freshwater spawning and incubation sites with water flow, quality and temperature conditions and substrate supporting spawning and incubation, (2) freshwater and estuarine migration corridors free of obstruction and with water flow, quality and temperature conditions supporting larval and adult mobility, and with abundant prey items supporting larval feeding after the yolk sac is depleted, and (3) nearshore and offshore marine foraging habitat with water quality and available prey, supporting juveniles and adult survival.

Atlantic sturgeon

Population structure. Atlantic sturgeon were once present in 38 river systems and, of these, spawned in 35 of them. Individuals are currently present in 36 rivers, and spawning occurs in at least 20 of these. Modern genetic analyses suggest that Atlantic sturgeon exhibit high fidelity to their natal rivers (Harwood 2010). Because of high natal river fidelity, it appears that most rivers support independent populations (Waldman and Wirgin 1998, Wirgin *et al.* 2000, King *et al.* 2001, Wirgin *et al.* 2002, Grunwald *et al.* 2008).

Distribution. Atlantic sturgeon once ranged from Hamilton Inlet on the coast of Labrador to the Saint Johns River in Florida and extralimitally to Bermuda and Venezuela (Smith and Clugston 1997, ASSRT 2007, Read 2010).

Reproduction and growth. The general life history pattern of Atlantic sturgeon is that of a long lived, late maturing, iteroparous, anadromous species.

Spawning intervals range from once every one to five years for males (Smith 1985, Bain 1997, Collins *et al.* 2000a, Schueller and Peterson 2010) and three to five years for females (Bain 1997, Stevenson and Secor 1999, Gales *et al.* 2010, Schueller and Peterson 2010). Fecundity increases with age and body size (ranging from 400,000 – 8 million eggs)(Van Eenennaam and Doroshov 1998, Dadswell 2006, Hammond 2010). The average age at which 50% of maximum lifetime egg production is achieved estimated to be 29 years, approximately 3-10 times longer than for other bony fish species examined (Boreman 1997).

Sturgeon eggs are highly adhesive and are deposited on the bottom substrate, usually on hard surfaces (e.g., cobble)(Gilbert 1989b, Smith and Clugston 1997). Hatching occurs approximately 94-140 hrs after egg deposition, and larvae assume a bottom-dwelling existence (Anonmyous 2010a). The yolksac larval stage is completed in about 8-12 days, during which time larvae move downstream to rearing grounds over a 6 – 12 day period (Kynard and Horgan 2002). During the daytime, larvae use benthic structure (e.g., gravel matrix) as refugia (Kynard and Horgan 2002). Juvenile sturgeon continue to move further downstream into brackish waters, and eventually become residents in estuarine waters for months or years.

Atlantic sturgeon may reach ages of 60 years or more, but aging studies are limited by inaccuracy once individuals are older than 15 years old (Rien and Beamesderfer 1994, Nakamoto *et al.* 1995a, Rossiter *et al.* 1995, Van Eenennaam *et al.* 1996, Stevenson and Secor 1999, Whiteman *et al.* 2004, Jackson *et al.* 2007). Individuals grow rapidly once they migrate out of natal streams, but experience slower growth once they reach sexual maturity and beyond (Harrison and Thurley 1974, Dovel and Berggren 1983). Individuals in southern waters may have shorter life spans.

Habitat. Estuaries along the coast that do not support Atlantic sturgeon spawning populations may still be important rearing habitats.

Movement. Atlantic sturgeon spawn in freshwater, but spend most of their sub-adult and adult life in the marine environment. While few specific spawning locations have been identified in the United States, through genetic analysis, many rivers are known to support reproducing populations. Early life stage Atlantic sturgeon coupled with upstream movements of adults suggest spawning adults generally migrate upriver in the spring and early summer; this includes February-March in southern systems, April-May in mid-Atlantic systems, and May-July in Canadian systems (Smith 1985, Bain 1997, Smith and Clugston 1997, Kahnle *et al.* 1998b). Some rivers may also support a fall spawning migration.

Sub-adult and adult Atlantic sturgeon undertake long marine migrations and utilize East Coast nearshore marine for rearing, feeding, and migrating (Harrison and Thurley 1974, Dovel and Berggren 1983, Bain 1997). Migratory sub-adults and adults normally occur in shallow (10-50m) waters dominated by gravel and sand substrate (Stein *et al.* 2004). Tagging and genetic data indicate that sub-adult and adult Atlantic sturgeon may travel widely after emigrating from rivers. Despite extensive mixing in coastal waters, Atlantic sturgeon display high site fidelity to their natal streams. Straying between rivers within a proposed DPS would sometimes exceed five migrants per generation, but between DPS exchanges usually less than one migrant per generation, with the exception of fish from the Delaware River straying more frequently to southern rivers (Grunwald *et al.* 2008).

Diet. Atlantic sturgeon feed primarily on polychaetes, isopods, and amphipods in the marine environment, while in fresh water, they feed on oligochaetes, gammarids, mollusks, insects, and chironomids (Moser and Ross 1995, Johnson *et al.* 1997, Haley 1998, Haley 1999, Brosse *et al.* 2002, Guilbard *et al.* 2007, Savoy 2007, Collins *et al.* 2008). There is some disagreement as to whether Atlantic sturgeon cease foraging during certain times or in certain places. Although there is some evidence to support a portion of individuals not foraging in spring or in freshwater, evidence also exists to support half to almost all individuals foraging in these circumstances (Brosse *et al.* 2002, Collins *et al.* 2008).

Several authors have found that polychaetes constitute a major portion of Atlantic sturgeon diets. Brosse *et al.* (2002) reported that over 90% of Atlantic sturgeon diet was polychaetes during spring, summer, and winter. Savoy (2007) found Atlantic sturgeon diets consisted of approximately 66% polychaetes and 27% decapods in Long Island Sound while at the mouth of the Connecticut River, individuals fed almost exclusively on polychaetes. At the mouth of the Hudson River, Haley (1999) found that sturgeon fed on 47% polychaetes, 27% amphipods, and 22% isopods. In North Carolina, Moser and Ross (1995) determined Atlantic sturgeon diets were different, feeding on 32% polychaetes, 28% isopods, 12% mollusks, and then other items. In South Carolina, Collins *et al.* (2008) identified the proportion of the sampled Atlantic sturgeon with each species in their guts and most guts contained polychaetes (over 50% of the fish that had been feeding had polychaetes in their guts).

Status and trends. On October 6, 2010, the NMFS proposed to list five DPSs of Atlantic sturgeon: the New York Bight, Chesapeake Bay, Carolina, and South Atlantic DPSs as endangered and the Gulf of Maine DPS as threatened (75 FR 61872 and 75 FR 61904). On June 6, 2011, the NMFS proposed protective measures for the Gulf of Maine DPS (76 FR 34023).

Prior to 1890, Atlantic sturgeon populations were at or near carrying capacity. In the mid-1800s, incidental catches of Atlantic sturgeon in the shad and river herring haul seine fisheries indicated that the species was very abundant (Armstrong and Hightower 2002). A major, targeted fishery did not exist until 1870 when a caviar market was established (Smith and Clugston 1997). Record landings were reported in 1890, where over 3350 metric tons (mt) of Atlantic sturgeon were landed from coastal rivers along the Atlantic Coast (Smith and Clugston 1997 , Matthiopoulos and Aarts 2010). Between 1890 and 1905, Atlantic sturgeon populations declined dramatically due to sale of meat and caviar. The majority of these landings (75%) were from the Delaware River fishery, which presumably supported the largest population along the Atlantic Coast (Matthiopoulos and Aarts 2010). Ten years after peak landings, the fishery collapsed in

1901, when less than 10% (295 mt) of its 1890 peak landings were reported. The landings continued to decline to about 5% of the peak until 1920 and remained between 1-5% thereafter. Between 1920 and 1998, the harvest level remained very low due to depleted populations. Prompted by research on juvenile production between 1985 and 1995 (Peterson *et al.* 2000), the Atlantic sturgeon fishery was closed by the Atlantic States Marine Fisheries Commission in 1998, when a coastwide fishing moratorium was imposed for 20 to 40 years, or at least until 20 year classes of mature female Atlantic sturgeon were present (ASMFC 1998).

Currently, the only populations that have been studied well enough to provide an estimate of size are from the Hudson and Altamaha Rivers. These two systems are considered the two largest spawning populations on the East Coast. Kahnle *et al.* (2007) reported that approximately 870 adults per year returned to the Hudson River between 1985 and 1995. Peterson *et al.* (2010) reported that approximately 324 and 386 adults per year returned to the Altamaha River in 2004 and 2005, respectively. Juvenile Atlantic sturgeon abundance may be a more precise way to measure the status of Atlantic sturgeon populations because it is believed that all age-1 and age-2 juveniles are restricted to their natal rivers (Dovel and Berggren 1983, Bain *et al.* 1999). Peterson *et al.* (2000) reported that there were approximately 4,300 age-1 and -2 Atlantic sturgeon in the Hudson River between 1985 and 1995. Schueller and Peterson (2010) reported that age-1 and -2 Atlantic sturgeon population densities ranged from 1,000 to 2,000 individuals over a 4 year period from 2004 to 2007.

The Hudson and Altamaha are presumed to be the healthiest populations within the US. Thus, other spawning populations within the US are predicted to have fewer than 300 adults spawning per year. However, evaluating the status of the species depends on the status of the smaller extant populations because maintaining those populations maintains genetic heterogeneity and having a broad range prevents a single catastrophic event from causing their extinction.

Natural threats. Naturally, these are small populations and in some rivers because of variable spawning returns, Allee affects could be an issue. During all stages of development, Atlantic sturgeon are sensitive to temperatures above 28°C (Niklitschek and Secor 2005, Anonmyous 2010b, Mcconnell *et al.* 2010) and dissolved oxygen levels below 4.3 to 4.7 parts per million (EPA 2003, Hindell *et al.* 2010, Taylor *et al.* 2010). Juvenile sturgeon are also stressed by high salinities until they mature and out migrate.

Anthropogenic threats. Factors likely play a larger role in this species' current status. Water quality, ship strikes, bycatch, dams, and poaching all contribute to the currently depressed populations of Atlantic sturgeon despite having very few natural predators.

The Atlantic Sturgeon Status Review Team (2007) determined Atlantic sturgeon in the Delaware River are at a moderately high risk of extinction because of ship strikes and sturgeon in the James River are at a moderate risk from ship strikes. Since that time, managers in the Hudson River are concerned that ship strikes may also be threatening Atlantic sturgeon populations there. In these systems, large ships move upstream from the mouths of the river to ports upstream through narrow shipping channels. The channels are dredged to the approximate depth of the ships, usually leaving less than 6 feet of clearance between the bottom of ships and the benthos of the river. Because of the size of the propellers used on large ships, everything along the bottom is sucked through the propellers. Large sturgeon are most often killed by ship strikes because smaller fish often pass through the propellers without making contact but larger sturgeon get hit. As shipping increases in the future, as has been predicted by the US Coast Guard, more Atlantic sturgeon are likely to be killed during encounters with ships. Besides the threats to Atlantic sturgeon from ships, the act of dredging the channel can also kill sturgeon. Dredging projects in the Kennebec, Delaware, James, Cape Fear, and Savannah Rivers put Atlantic sturgeon at moderate risk (ASSRT 2007). Dredging primarily affects sturgeon by removing food resources and homogenizing habitat, eliminating holding areas and other high quality habitat. Also, sometimes Atlantic sturgeon are attracted to the sediment plume created during dredging operations and are killed by the dredge itself.

Atlantic sturgeon are caught as bycatch in several fisheries both within river systems and along the coast. In the James River, bycatch in the striped bass fishery poses a moderately high risk to the species, while it poses a moderate risk in nearly every other river system on the East Coast (ASSRT 2007). While these determinations were made for Atlantic sturgeon in each river system, the majority of the commercial fisheries interactions occur in estuaries and along the coast, where sturgeon from all rivers could be captured as bycatch.

On the East Coast, there is no good means of fish passage for Atlantic sturgeon in the systems with dams. Furthermore, as human populations grow along the Atlantic Coast and droughts were common over the past decade, it is likely that many more rivers on the East Coast could be dammed. Sturgeon in the Santee-Cooper River system and the Cape Fear River are at a moderately high risk because of dams. Additionally, sturgeon in the Neuse River are at a moderate risk from dams.

Atlantic sturgeon particularly were overfished during the late 1880s, peaking in 1890 and the fishery collapsed in 1901 (Jehl and Cooper 1980). While the fishery remained open following the initial peak harvest period, landings remained low through the 20th century until 1996 when the fishery was closed due to concerns about the recovery of their populations.

Atlantic sturgeon have also been impacted by industrialization, poor water quality, and loss of habitat (Van Eenennaam *et al.* 1996, Jager *et al.* 2001, Collins *et al.* 2002, Stein *et al.* 2004). Most Atlantic sturgeon managers and researchers consider water quality as a moderate risk to every DPS in the United States (ASSRT 2007). Atlantic sturgeon are sensitive to pesticides, heavy metals, and other toxins in the aquatic environment.

Critical habitat. Critical habitat has not been proposed for Atlantic sturgeon.

Marine invertebrates

Elkhorn coral

Description of the species. Although they resemble plants, elkhorn coral is a colony of small shelled animals that collaboratively form frond-like branches radiating from a central trunk that is firmly attached to the sea floor. The largest species of its genus, colonies can reach at least 6.6 feet high and 13 feet in diameter (Veron 2000). Corallites (branches of radial arms of calcium carbonate) are tube-like and porous, 0.08 inch to 0.16 inch long, about 0.08 inch in diameter, white near the growing tip, and brown to tan away proximally.

Distribution. Elkhorn coral is found widely in the Caribbean, including in the Florida Keys, Abaco Island (The Bahamas), Alacran Reef, Mexico, Belize, Colombia, Costa Rica, Guatemala, Honduras, Nicaragua, Panama, Venezuela, Bonaire, Cayman Islands, Jamaica, Puerto Rico, US Virgin Islands, Navassa, and throughout the West Indies (Goreau 1959, Kornicker and Boyd 1962, Storr 1964, Scatterday 1974, Jaap 1984, Dustan and Halas 1987, NMFS 2006e). However, abundance within the distribution is reduced, largely due to water temperature and quality issues.

Growth and reproduction. Elkhorn corals employ both sexual and asexual reproduction. Sexual reproduction is accomplished by releasing sperm and egg during spawning events. Colonies are referred to as simultaneous hermaphrodites, meaning that a given colony contains both female and male reproductive sex organs (Szmant 1986). Spawning events are relatively short, with gametes released only a few nights during July, August, and/or September. In some populations, spawning is synchronous after a full moon. Annual egg production in Puerto Rico was estimated to be 3,870 to 5,100 eggs per square inch of living coral tissue (Szmant 1986). Once fertilization occurs, planktonic larvae form before settling and metamorphosizing on appropriate substrates, preferably coralline algae (Bak 1977, Sammarco 1980, Rylaarsdam 1983). Initial calcification ensues and develop into daughter corallites.

Studies indicate that larger colonies (as measured by surface area of the live colony) have higher fertility and fecundity rates; over 80% of the colonies larger than 620 inches^2 were fertile. Estimated colony size at sexual maturity was 248 inches^2 and the smallest reproductive colony observed was 6.3 inches^2 by 3.15 inches^2 (Soong and Lang 1992).

Biological and physical factors affect spatial and temporal patterns of recruitment. These include substrate availability and community structure, grazing pressure, fecundity, mode and timing of reproduction, behavior of larvae, hurricane disturbance, physical oceanography, the structure of established coral assemblages, and chemical cues (Lewis 1974, Birkeland 1977, Goreau *et al.* 1981, Rogers *et al.* 1984, Baggett and Bright 1985, Harriott 1985, Hughes and Jackson 1985, Sammarco 1985, Morse *et al.* 1988, Fisk and Harriott 1990, Richmond and Hunter 1990).

Growth rates are relatively rapid, expressed as the linear extension of branches, ranging from 1.57 to 4.33 inches annually, and have enabled elkhorn coral to construct significant reefs in several locations throughout the Caribbean (Vaughan 1915, Jaap 1974, Adey 1978). Branching species, such as acroporid corals, grow differentially in response to light such that coral polyp growth maximizes exposure to available light (Kaniewska *et al.* 2009). Growth can also occur from fragmentation and dispersal (Tunnicliffe 1981, Bak and Criens 1982). A broken branch may be carried by waves and currents to another location and, if favorable, branches grow into a new colony. Rapid growth and fragment dispersal facilitate a competitive advantage for elkhorn coral relative to other coral and benthic species (Shinn 1976, Neigel and Avise 1983, Jaap *et al.* 1989).

Elkhorn coral require relatively clear water and depend almost entirely upon symbiotic photosynthesizers (zoozanthelle) for nourishment (Porter 1976, Lewis 1977, Jaap *et al.* 1989, Mieog *et al.* 2009) and is much more susceptible to increases in water turbidity than are some other corals. Different strains of symbiotic zoozanthelle (*Symbiodinium* spp.) can confer different thermal and light tolerances to acroporiids (Abrego *et al.* 2009, Ainsworth and Hoegh-Guldberg 2009, Abrego *et al.* 2010). The type of *Symbiodinium* spp. may change during ontogeny or remain the same, depending upon acroporiid species, and may be the same as parent colonies or not (Baird *et al.* 2007, Gómez-Cabrera *et al.* 2008, Abrego *et al.* 2009).

Habitat. Colonies of elkhorn coral often grow in dense stands and form interlocking framework known as thickets in fringing and barrier reefs, ranging in depth from 3.3 to 49 feet (Jaap 1984, Dustan 1985, Dustan and Halas 1987, Tomascik and Sander 1987, Wheaton and Jaap 1988). However, optimal depth range is considered to be 3.3 to 16.4 feet in depth, with possible exposure at low tide (Goreau and Wells 1967). Colonies generally do not form thickets below 16.4 feet, with maximum water depths of framework construction ranging from 10 to 39.4 feet (Lighty *et al.* 1982). Elkhorn coral thrive in shallow reef zones where wave energy is a significant factor. In areas with strong wave energy conditions only isolated colonies occur, while denser thickets may develop in intermediate wave energy conditions (Geister 1977). The preferred habitat of elkhorn coral is the seaward face of a reef (Shinn 1963, Cairns 1982, Rogers *et al.* 1982).

Status and trends. Elkhorn coral was listed as threatened under the ESA on May 9, 2006 (71 FR 26852). Elkhorn coral underwent precipitous declines in the early 1980s throughout its range and this decline has continued. Although quantitative data on historical distribution and abundance are scarce, best available data indicate declines in abundance (coverage and colony numbers) by greater than 97%. Recovery from a bleaching event in 2005 is expected to take 10-12 years; this is after a previous event in 1997. In all, roughly one-third of the *Acropora palamata* genotypes have been lost as a result of these events (Miller and Williamson 2010).

Natural threats. The overriding threats are disease, temperature-induced “bleaching” (loss of zoozanthelle), and physical damage from hurricanes (Carpenter *et al.* 2008, Mallela and Crabbe 2009, Baskett *et al.* 2010). Disease is widespread, episodic, and unpredictable in its occurrence and results in high mortality. This is primarily due to a disassociation of zoozanthelle from coral tissue. Just prior to this, coral epithelium and gastrodermis tissue begins to decay and die, likely as a result of stress to the individual coral (Ainsworth *et al.* 2008). Optimal water temperatures range from 77° to 84°F, with mortality observed at 61° and 96°F (Jaap 1979, Roberts *et al.* 1982). High light levels can also induce mortality. Synergistic analyses have found that high temperature increases the risk of colony mortality under a variety of sediment loading conditions, but excessive sediment appears to reduce mortality risk under high light and temperature regimes, possibly by reducing exposure to these stressors (Anthony *et al.* 2007, Boyett *et al.* 2007). High sediment with otherwise good light and temperature conditions appears to increase colony mortality (Anthony *et al.* 2007). Elkhorn coral require near oceanic salinities (34 to 37 parts per thousand). High temperature or rapid heating can result in heat shock and alter cellular metabolism within the coral as well as possibly hinder immune response or the ability of zoozanthelle to thrive (Rodriguez-Lanetty *et al.* 2009, Middlebrook *et al.* 2010). Bleaching can occur due to adverse environmental conditions (Ghiold and Smith 1990, Williams and Bunkley-Williams 1990) and is currently a significant factor in deteriorating coral reef health. In 2005, wide-scale bleaching occurred throughout the Caribbean with wide-scale mortality, with some areas reaching 95% of coral colonies affected (Wilkinson and Souter 2008). The US Virgin

Islands, a location of *Acropora* critical habitat, experienced greater than 50% mortality of corals, the greatest level ever recorded. Puerto Rico and Florida (additional areas of *Acropora* critical habitat) also experienced disease rates of 50% of coral colonies or greater. Bleaching was associated with unusually warm waters in the region. Encouragingly, bleaching events can lead to increased thermal tolerance in affected reefs, meaning that subsequent bleaching events are not as severe (Maynard *et al.* 2008). A record number of hurricanes also caused extensive damage to coral reefs; the prevalence of hurricanes and subsequent coral reef damage has been linked to climate change (Wilkinson and Souter 2008). Ocean acidification is also a threat due to the increased solubility of calcium carbonate in even slightly more acidic sea water (thereby eroding the shells which form coral hard parts)(Anthony *et al.* 2008, De'ath *et al.* 2009, Wei *et al.* 2009, Crawley *et al.* 2010). Acidification also reduces the thermal tolerance of corals, meaning that bleaching can occur at lower temperatures (Anthony *et al.* 2008). Hurricanes can cause wide-scale inhibition of recruitment in years following storm passage as well as physical damage to coral colonies themselves (Mallela and Crabbe 2009).

Anthropogenic threats. Threats to elkhorn coral also include eutrophication, sedimentation, anchoring, which degrade coral condition and increase synergistic stress effects (e.g. bleaching).

Critical habitat. NMFS published a final rule to designate critical habitat for elkhorn and staghorn corals on November 26, 2008 (73 FR 72210). This habitat serves as substrate of suitable quality and availability, in water depths from the mean high water line to 98 feet (except along some areas of Florida, where 6 foot contour is the shoreward limit), to support successful larval settlement, recruitment, and reattachment of fragments. Four specific areas are proposed for designation: the Florida unit, which comprises approximately 1,329 square miles of marine habitat; the Puerto Rico unit, which comprises approximately 1,383 square miles of marine habitat; the St. John/St. Thomas unit, which comprises approximately 121 square miles of marine habitat; and the St. Croix unit, which comprises approximately 126 square miles of marine habitat. NMFS proposes to exclude one military site, comprising approximately 47 square miles, because of national security impacts. The lone PCE identified thus far is natural consolidated hard substrate or dead coral skeleton that are free from fleshy or turf macroalgae cover and sediment cover. This feature is essential to the conservation of these two species because of the extremely limited recruitment currently being observed and the need for this species to have habitat to recruit into.

Staghorn coral

Description of the species. Although they resemble plants, staghorn coral is a colony of small shelled animals that collaboratively form staghorn-antler-like colonies, with cylindrical, straight or slightly curved diverging branches. Branches are 0.1 inch to 0.6 inch in diameter and rarely may grow back together. Colonies in turbulent water are smaller than in calm water, with greater branch density. Branching is irregular and secondary branches form at 60 to 90 degree angles relative to a primary branch. Prominent axial corallites (branches of radial arms of calcium carbonate) form at branch tips; bract-like corallites radiate symmetrically around branches. Individual colonies are up to 5 feet across and typically form monospecific thickets. Tissue color ranges from golden yellow to medium brown, with little or no color near the growing branch tips. The colony may or may not be firmly attached to the sea floor.

Distribution. Staghorn coral is found widely in the Caribbean, including in the Florida Keys, Abaco Island (The Bahamas), Alacran Reef, Mexico, Belize, Colombia, Costa Rica, Guatemala, Honduras, Nicaragua, Panama, Venezuela, Bonaire, Cayman Islands, Jamaica, Puerto Rico, US Virgin Islands, Navassa, and throughout the West Indies (Goreau 1959, Kornicker and Boyd 1962, Storr 1964, Scatterday 1974, Jaap 1984, Dustan and Halas 1987, NMFS 2006e). However, abundance within the distribution is reduced, largely due to water temperature and quality issues.

Growth and reproduction. Staghorn corals employ both sexual and asexual reproduction. Sexual reproduction is accomplished by releasing sperm and egg during spawning events (Szmant 1986). Colonies are referred to as simultaneous hermaphrodites, meaning that a given colony contains both female and male reproductive sex organs. Spawning events are relatively short, with gametes released only a few nights during July, August, and/or September. In some populations, spawning is synchronous after a full moon. Annual egg production in Puerto Rico was estimated to be 3,870 to 5,100 eggs per square inch of living coral tissue (Szmant 1986). Once fertilization occurs, planktonic larvae form before settling and metamorphosizing on appropriate substrates, preferably coralline algae (Bak 1977, Sammarco 1980, Rylaarsdam 1983). Initial calcification ensues and develop into daughter corallites.

Studies indicate that larger colonies (as measured by surface area of the live colony) have higher fertility and fecundity rates; colonies with a branch length longer than 3.5 inches were fertile and over 80% of colonies with branches longer than 6.7 inches were fertile (Soong and Lang 1992). Estimated size at sexual maturity is 6.7 inches in branch length and the smallest known reproductive colony was 3.5 inches in branch length (Soong and Lang 1992).

Biological and physical factors affect spatial and temporal patterns of recruitment. These include substrate availability and community structure, grazing pressure, fecundity, mode and timing of reproduction, behavior of larvae, hurricane disturbance, physical oceanography, the structure of established coral assemblages, and chemical cues (Lewis 1974, Birkeland 1977, Goreau *et al.* 1981, Rogers *et al.* 1984, Baggett and Bright 1985, Harriott 1985, Hughes and Jackson 1985, Sammarco 1985, Morse *et al.* 1988, Fisk and Harriott 1990, Richmond and Hunter 1990).

Growth rates are relatively rapid, expressed as the linear extension of branches, ranging from 1.2 to 4.3 inches annually, and have enabled staghorn coral to construct significant reefs in several locations throughout the Caribbean (Vaughan 1915, Jaap 1974, Adey 1978). Branching species, such as acroporid corals, grow differentially in response to light such that coral polyp growth maximizes exposure to available light (Kaniewska *et al.* 2009). During the 1970s there were vast fields, or thickets, of staghorn coral on many reefs. The nominal situation in 2004 was isolated branches and small thickets, 1.6 to 3.3 feet across. Growth can also occur from fragmentation and dispersal (Tunnicliffe 1981, Bak and Criens 1982). A broken branch may be carried by waves and currents to another location and, if favorable, branches grow into a new colony. Rapid growth and fragment dispersal facilitate a competitive advantage for staghorn coral relative to other coral and benthic species (Shinn 1976, Neigel and Avise 1983, Jaap *et al.* 1989). Larval recruitment is influenced by the type and availability of benthic substrate, with certain types of coral or rock substrates resulting in greater or lesser recruitment success (Ritson-Williams *et al.* 2009).

Habitat. Historically, staghorn coral so dominated reef systems within the 23 to 49 feet depth that the area became known as the staghorn zone. In other reef systems (Jamaica, Cayman Islands, Belize, and eastern Yucatan), staghorn coral was a major mid-depth (33 to 82 feet) reef-builder (Adey 1977, 1978). Historically, staghorn coral was reported from depths ranging from surface to 200 feet, although it is considered rare below 66 feet (Goreau and Goreau 1973). In southeastern Florida, this species historically occurred on the outer reef (52 to 66 feet), on spur, groove bank, and transitional reefs, and on octocoral-dominated hard-bottom (Goldberg 1973, Davis 1982, Jaap 1984, Wheaton and Jaap 1988). Colonies were common in back- and patch-reef habitats (Gilmore and Hall 1976, Cairns 1982). Although staghorn coral colonies are sometimes found interspersed among colonies of elkhorn coral, they are generally in deeper water or seaward of the elkhorn zone and more protected from wave action.

Staghorn coral require relatively clear water and depend almost entirely upon symbiotic photosynthesizers (zoozanthelle) for nourishment (Porter 1976, Lewis 1977, Jaap *et al.* 1989, Mieog *et al.* 2009) and is much more susceptible to increases in water turbidity than are some other corals. Different strains of symbiotic zoozanthelle (*Symbiodinium* spp.) can confer different thermal and light tolerances to acroporiids (Abrego *et al.* 2009, Ainsworth and Hoegh-Guldberg 2009, Abrego *et al.* 2010). The type of *Symbiodinium* spp. may change during ontogeny or remain the same, depending upon acroporiid species, and may be the same as parent colonies or not (Baird *et al.* 2007, Gómez-Cabrera *et al.* 2008, Abrego *et al.* 2009).

Status and trends. Staghorn coral was listed as threatened under the ESA on May 9, 2006 (71 FR 26852). Staghorn coral underwent precipitous declines in the early 1980s throughout its range and this decline has continued. Although quantitative data on historical distribution and abundance are scarce, best available data indicate declines in abundance (coverage and colony numbers) by greater than 97% (Figure 23).

Staghorn corals still occupy their historic range, but localized range reductions and extirpations have occurred with most populations experiencing losses from 80-98% of their 1970s baseline (Bruckner 2002). Monitoring data from around the USVI indicates that staghorn corals have virtually disappeared from the north side of Buck Island, St. Croix, and only a few localized areas off the southern reef contain staghorn corals, representing 2-3% of the coral cover in these areas (Rogers *et al.* 2002). Surveys of fragments of staghorn from nearshore areas of St. Thomas and outlaying cays indicate that colonies of these corals were once much more abundant than the numbers recorded in the 2003 survey (Rogers *et al.* 2008).

In Puerto Rico, well-developed and dense thickets of staghorn coral were present through the late 1970s at many reefs surrounding the main island, and also the offshore islands of Mona, Vieques and Culebra (Almy and Carrión-Torres 1963, McKenzie and Benton 1972, Goenaga and Cintrón 1979, Boulon Jr. 1980). Later, in 1978-79 during an island-wide survey, staghorn coral was found on only 20% of those reefs (Bruckner 2002). Prior to Hurricane David in 1979, 20 random 0.6 m^2 photoquadrats were selected from each of 10, 40-m-long transects parallel to the depth contours across the reef (16.7 to 19.2 m depth). Based on analysis of point count data, staghorn coral had a mean total cover of 31.1% (range of 9.9 to 56.9%); after the storm, total cover of staghorn coral dropped to a mean of 0.90% (range of 0.02 to 2.7%)(NMFS 2008a). In the summer of 2004, there was an epidemic outbreak of white pox disease at Los Corchos Reef in Culebra, Puerto Rico. Prior to the outbreak, coral cover on the reef reached values of 80%. However, three weeks after Tropical Storm Jeanne, 80 to 90% of the staghorn coral colonies at permanent monitoring sites at Los Corchos were already dead or dying; likely as a result of impacts from both disease and storm damage (NMFS 2008a). During the 2005 bleaching event, near Culebra Island, almost 100% of staghorn colonies suffered partial to complete mortality due to bleaching (García-Saís *et al.* 2008). Similar to the situation in USVI, the bleaching event was followed by a white plague-like massive outbreak that caused mass mortality and resulted in a net 20-60% decline in living coral cover at surveyed reefs of the east coast within a period of approximately six months.

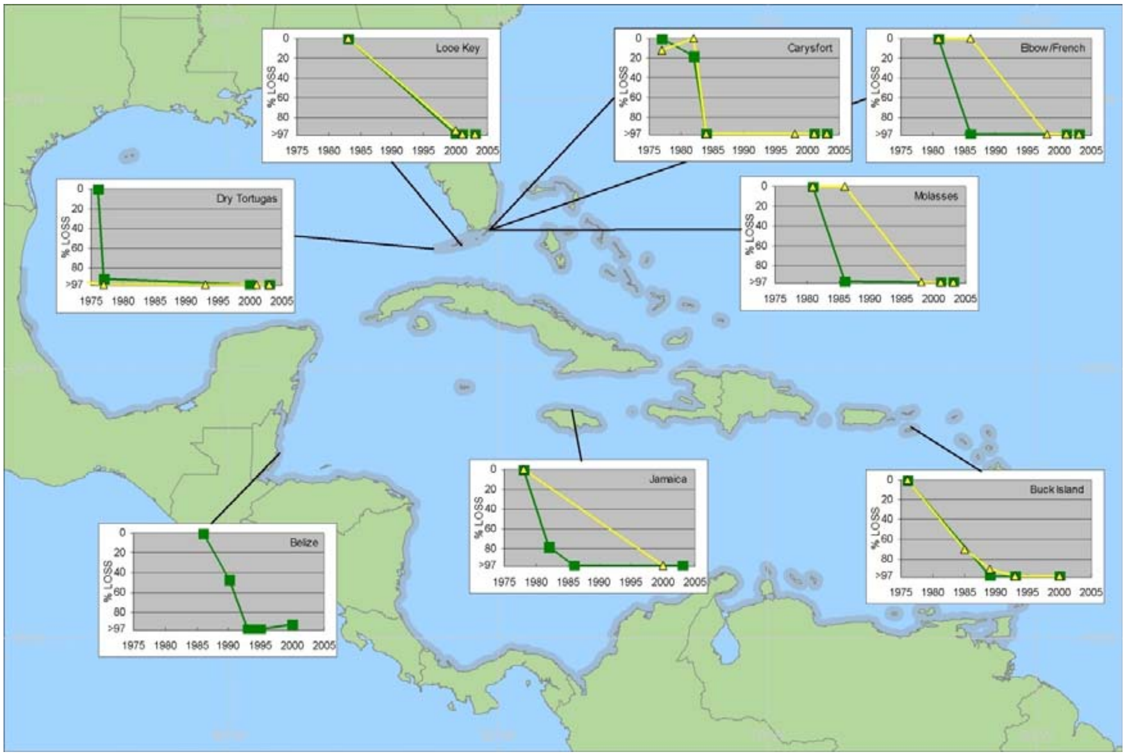

Figure 23. Percent loss of staghorn and elkhorn coral throughout the Caribbean Sea.

Following the 2005 bleaching event, monitoring data indicate that total coral cover is now less than 12% on many reefs (Rogers *et al.* 2008). Coral mortality due to the 2005 bleaching event was more severe than at any time in the last 40 years of monitoring in USVI (Woody *et al.* 2008). Staghorn corals suffered widespread mortality associated with the 2005 bleaching event and current monitoring data does not indicate significant recovery (Rothenberger *et al.* 2008, Woody *et al.* 2008). Overall, colonies of Atlantic *Acropora* have declined by up to 98% and live colonies were no longer present at many study sites in the USVI following the 2005-2006 bleaching event.

Natural threats. The overriding threats are disease, temperature-induced "bleaching" (loss of zoozanthelle), and physical damage from hurricanes (Carpenter *et al.* 2008, Mallela and Crabbe 2009, Baskett *et al.* 2010). Disease is widespread, episodic, and unpredictable in its occurrence and results in high mortality. This is primarily due to a disassociation of zoozanthelle from coral tissue. Just prior to this, coral epithelium and gastrodermis tissue begins to decay and die, likely as a result of stress to the individual coral (Ainsworth *et al.* 2008). Optimal water temperatures range from 77º to 84ºF, with mortality observed at 61º and 96ºF (Jaap 1979, Roberts *et al.* 1982). High light levels can also induce mortality. Synergistic analyses have found that high temperature increases the risk of colony mortality under a variety of sediment loading conditions, but excessive sediment appears to reduce mortality risk under high light and temperature regimes, possibly by reducing exposure to these stressors (Anthony *et al.* 2007, Boyett *et al.* 2007). High sediment with otherwise good light and temperature conditions appears to increase colony mortality (Anthony *et al.* 2007). Elkhorn coral require near oceanic salinities (34 to 37 parts per thousand). High temperature or rapid heating can result in heat shock and alter cellular metabolism within the coral as well as possibly hinder immune response or the ability of zoozanthelle to thrive (Rodriguez-Lanetty *et al.* 2009, Middlebrook *et al.* 2010). Bleaching can occur due to adverse environmental conditions (Ghiold and Smith 1990, Williams and Bunkley-Williams 1990) and is currently a significant factor in deteriorating coral reef health. The major El Niño/La Niña Southern Oscillation cycle in 1997-1998 resulted in a large bleaching event in the Caribbean and the Atlantic, as well as massive losses of corals in the Indian Ocean and Western Pacific (Wilkinson and Souter 2008). However, the most significant bleaching event to date in the USVI and other areas of the Caribbean occurred in 2005 when sea surface temperatures exceeded the 29.5°C coral bleaching threshold for twelve weeks, and maximum temperatures exceeded 30°C (Woody *et al.* 2008). Bleaching occurred in twenty-two species, including *Acropora*, over a wide range of depths and affected more than 90% of the coral cover, on average, between July and November in the USVI (Woody *et al.* 2008). Wide-scale mortality, with some areas reaching 95% of coral colonies affected, resulted from this event (Wilkinson and Souter 2008). The US Virgin Islands, a location of *Acropora* critical habitat, experienced greater than 50% mortality of corals, the greatest level ever recorded. Puerto Rico and Florida (additional areas of *Acropora* critical habitat) also experienced disease rates of 50% of coral colonies or greater. Bleaching was associated with unusually warm waters in the region. Encouragingly, bleaching events can lead to increased thermal tolerance in affected reefs, meaning that subsequent bleaching events are not as severe (Maynard *et al.* 2008). A record number of hurricanes also caused extensive damage to coral reefs; the prevalence of hurricanes and subsequent coral reef damage has been linked to climate change (Wilkinson and Souter 2008). Ocean acidification is also a threat due to the increased solubility of calcium carbonate in even slightly more acidic sea water (thereby eroding the shells which form coral hard parts)(Anthony *et al.* 2008, De'ath *et al.* 2009, Wei *et al.* 2009, Crawley *et al.* 2010). Acidification also reduces the thermal tolerance of corals, meaning that bleaching can occur at lower temperatures (Anthony *et al.* 2008). Hurricanes can cause wide-scale inhibition of recruitment in years following storm passage as well as physical damage to coral colonies themselves (Mallela and Crabbe 2009).

White band disease is thought to be the major factor responsible for the rapid loss of Atlantic *Acropora* due to mass mortalities. White band disease is the only coral disease to date that has been documented to cause major changes in the composition and structure of reefs (Humann and Deloach 2003). In 2011, Sutherland *et al.* (2011) were able to definitively identify human waste as a cause for white pox disease in elkhorn corals.

Reductions in long-term water clarity can also reduce the coral photosynthesis to respiration ratio (P/R ratio). Telescnicki and Goldberg (1995) and Yentsch *et al.* (2002) found that elevated turbidity levels did not affect gross photosynthetic oxygen production, but did lead to increased respiration that consumed the products of photosynthesis with little remaining for coral growth.

Unfortunately, since staghorn corals are broadcast spawners once colonies become rare, the distance between colonies may limit fertilization success and there is substantial evidence to suggest that sexual recruitment of staghorn corals is currently compromised. Reduced colony density in some areas is compounded by low genotypic diversity, indicating that fertilization success and consequently, larval availability, is likely reduced. This can have long-term implications for genetic variability of remaining colonies due to the reduced potential for exchange of genetic material between populations that are spatially further apart (Bruckner 2002).

Data on levels of genetic diversity and population structure suggest that there is a population structure among islands, and even over spatial scales of no more than 20 km, as well as varying degrees of genetic diversity within local populations (Lirman 2002, Vollmer 2002). For instance, one clone of staghorn coral may dominate areas up to 10 m^2 in size and the clones are generally spatially discrete with larval exchange between staghorn populations as close as 2 to 15 km being extremely limited, suggesting that larval sources need to be conserved on a very small spatial scale (Baums *et al.* 2005, Vollmer and Palumbi 2007).

Anthropogenic threats. Threats to staghorn coral are exacerbated further by eutrophication, sedimentation, and anchoring, which degrade coral condition and increase synergistic stress effects (e.g. bleaching). Excessive sedimentation can smother corals and increased nutrient availability promotes algal growth on corals, leading to light blockage to zoozanthelle and death of corals (*Acropora* Biological Review Team 2005). Although reefs in the Florida Keys currently experience about 10% macroalgal cover or less, much of the wider Caribbean Sea may exceed 20% cover (Bruno 2008), inhibiting and reducing coral survival. Global warming is also projected to have negative impacts on coral survival through coral bleaching, increased storm intensity, and reduced calcification (*Acropora* Biological Review Team 2005).

Critical habitat. NMFS published a final rule to designate critical habitat for elkhorn and staghorn corals on November 26, 2008 (73 FR 72210). This habitat serves as substrate of suitable quality and availability, in water depths from the mean high water line to 98 feet (except along some areas of Florida, where 6 foot contour is the shoreward limit), to support successful larval settlement, recruitment, and reattachment of fragments. Four specific areas are proposed for designation: the Florida unit, which comprises approximately 1,329 square miles of marine habitat; the Puerto Rico unit, which comprises approximately 1,383 square miles of marine habitat; the St. John/St. Thomas unit, which comprises approximately 121 square miles of marine habitat; and the St. Croix unit, which comprises approximately 126 square miles of marine habitat. NMFS proposes to exclude one military site, comprising approximately 47 square miles, because of national security impacts. The lone PCE identified thus far is natural consolidated hard substrate or dead coral skeleton that are free from fleshy or turf macroalgae cover and sediment cover. This feature is essential to the conservation of these two species because of the extremely limited recruitment currently being observed and the need for this species to have habitat to recruit into.

White abalone

Distribution. White abalone occur along the US west coast among offshore islands and banks (particularly Santa Catalina and San Clemente islands) and mainland inshore waters from Point Conception, California south to Punta Abreojos, Baja California, Mexico (Bartsch 1940, Cox 1960, 1962). White abalone occur primarily along the mainland coast in their northern and southern range, but are more frequently at the offshore islands (especially San Clemente and Santa Catalina islands) in the middle portion of the California range (Cox 1962, Leighton 1972). However, individuals have also been found around several Mexican islands including Isla Cedros and Isla Natividad (Guzmán Del Proó 1992). There are no recognized subspecies of white abalone although there is one possible subspecies of white abalone inhabiting Guadalupe Island, Mexico (Hobday and Tegner 2000).

Habitat. White abalone occupy kelp forests in relatively exposed areas of low-relief rocky habitat surrounded by sand. Large juvenile and small adult abalone (3 to 4 inches) are cryptic and seek shelter in crevices and under rocks before, as adults, moving to more open habitats on the tops and sides of rocks where food is more plentiful (Haaker *et al.* 1986, Hobday and Tegner 2000). Adult white abalone are found most abundantly at depths of 82 to 99 feet, but may occur in waters from 66 to 197 feet (Hobday and Tegner 2000).

Feeding. White abalone eat algae, feeding as postlarvae and early juveniles on bacteria, sessile pennate diatoms, and other benthic microflora. As advanced juveniles and adults, drifting brown algae and microalgal films provide the primary source of nutrition (Tutschulte 1976).

Growth and reproduction. Abalone aggregate for spawning, but low numbers and physical barriers can prevent large spawning aggregations from forming (Babcock and Keesing 1999, Leet *et al.* 2001). A brief annual spawning event occurs *en mass* generally between February and April (Tutschulte 1976). Although an average female is capable of producing over 20 million larvae over her lifetime, larval survival to adulthood is estimated at <1% (Leighton 2000).

Twenty four hours after fertilization, a free-swimming larva emerges from the fertilized egg and joins the plankton (Leighton 1989, 2000). After 2 to 3 weeks in the plankton the larvae settle to the bottom. One to 3 months after settlement juveniles are fully formed and resemble adults. After 2 to 4 years, white abalone are mature and inhabit the tops and sides of rocky substrates. However, low food resource availability has been shown to stunt growth in blacklip abalone (Saunders *et al.* 2009a, Saunders *et al.* 2009b).

Movement. Abalone movements tend to vary by individual, with some staying at a particular home site and others undertaking extensive movements (Momma and Sato 1969, Ault and Martini 1987). This has been further characterized in green lip abalone by Cennie *et al.* (2009a), where individuals belonged to either a sedentary or a wandering clade (which moved over larger areas and covered greater distances, occupied inner portions of hides more frequently, responded more rapidly to food odor, foraged for longer periods, and displaced conspecifics from food patches).

Life span. Researchers believe that abalone are long-lived, but the average life span is unclear

(Hobday and Tegner 2000). Models of growth predict maximum size is reached in 34 to 35 years and a maximum life span of 35 to 40 years has been estimated (Tutschulte 1976, Tutschulte and Connell 1988).

Status and trends. On May 29, 2001, the white abalone was listed as an endangered species throughout its range under the ESA (66 FR 29046). In the white abalone status review, Hobday and Tegner (2000) estimated pre-exploitation abundance at 2,221,800 abalone, but the population in 1996 to 1997 was estimated at 1,613 individuals, representing a 99.9% decline, and was estimated to disappear by 2010. White abalone recovery is hindered by low spawning densities, resulting in recruitment failure. White abalone along Mexico are believed to be depleted, but their status is generally unknown. Based upon survey data, Hobday *et al.* (2001) updated the 1996 to 1997 white abalone abundance estimate to 2,540.

White abalone recruitment is highly variable (Tutschulte 1976). However, estimates of population size have been difficult to calculate because estimates are only based upon adults, as juveniles are infrequently observed. White abalone observed during surveys were of large size which corresponds to predicted ages near the end of the predicted life span (Davis 1996, Davis *et al.* 1998, Hobday and Tegner 2000, Hobday *et al.* 2001). Because no white abalone were observed in the smaller age/size classes during the surveys there appears to be a lack of successful recruitment since the 1960s (Hobday and Tegner 2000).

Natural threats. Natural pressures exist from sea otter predation (Johnson *et al.* 2009) and the low density of individuals during spawning events. Interspecific competition has not been studied in white abalone, but blacklip abalone face significant competition for algal resources by sea urchins (Strain and Johnson 2009).

Anthropogenic threats. White abalone numbers were severely reduced due to excessive harvest. This has led to below-threshold spawning densities in many areas that are blamed for the inability of the species to recover. Although small-scale aquaculture takes have occurred to attempt captive breeding and recovery, these takes are small in number. Otherwise, substantial human harvesting is not known. No commercial or recreational takes are permitted under ESA protection.

Although toxicology of abalone is poorly known, red abalone have demonstrated metabolic breakdown when exposed to the pesticide 3-trifluoromethyl-4-nitrophenol (Viant *et al.* 2001). Silver, cadmium, and mercury are also known to bioaccumulate in abalone, likely from ingested algae (Huang *et al.* 2008). Update rates have been measured as 1.78 L g^{-1} d^{-1} for silver, 0.056 L g^{-1} d^{-1} for cadmium and 0.32 L g^{-1} d^{-1} for mercury, of which 58 to 83%, 33 to 59%, and 65 to 78%, respectively, is assimilated. Abalone are known to bioaccumulate high levels of heavy metals in the presence of high environmental concentrations (Wang *et al.* 2009).

Changes in sea surface temperatures have been suggested as a driving force in altering red abalone distribution in the past (Braje *et al.* 2009); it is unknown what affect, if any climate change may have on white abalone.

Critical habitat. Critical habitat has not been designated for white abalone.

Black abalone

Distribution. Black abalone historically occurred between Coos Bay, Oregon, and Cape San Lucas, Baja California (Cox 1962), but were rare north of San Francisco (Morris *et al.* 1980). Distribution extended to the Channel Islands and from Cedros to Punta Asuncion along Baja, Mexico (Guzmán Del Proó 1992). Present occurrence remains throughout much of this range, although greatly reduced.

Growth and reproduction. Spawning occurs during spring and summer (Cox 1960). Synchronicity of gamete release is vital, as likelihood of fertilization is reliant upon dense adult aggregation and subsequent high egg and sperm density (Davis 1996). Fecundity increases exponentially with size, with small mature females producing a few hundred thousand eggs each year, but older individuals producing 10 to 15 million eggs (Hahn 1989). Fertilized eggs sink and hatch into free-swimming larvae within 72 hours. After one or two weeks, larvae settle (CDFG 1993). Larval mortality is assumed to be high (Leighton 1972). At 1.5 inches (roughly 3 years of age), black abalone are considered sexually mature (Blecha *et al.* 1992). Size at sexual maturity appears to decrease with latitude, suggesting precocity in southern portions of the range (Munoz and Camacho 1976, Guzman del Proo *et al.* 1980).

Feeding. Black abalone eat algae, with larvae eating pelagic plankton and postlarvae feeding upon on bacterial films, benthic diatoms (Cox 1962, Ault 1985), and coralline algae. Juveniles and adults feed upon large-bodied algae, such as giant and feather-boa kelp (Cox 1962, Howorth 1978). However, low food resource availability has been shown to stunt growth in blacklip abalone (Saunders *et al.* 2009a, Saunders *et al.* 2009b).

Habitat. Black abalone occupy rocky intertidal and shallow subtidal zones (Haaker *et al.* 1986). Depth distribution is usually from the shore to about 10 feet of water depth. Mobility patterns are size dependent (Blecha *et al.* 1992). Individuals above 4.1 inches undergo limited movement in exposed rocky locations. Juveniles (smaller than 1.6 inches) remain within rocky crevices during daylight hours, but become more active at night (Cox 1960). At 3 to 4 inches in length, when they emerge into more open rocky habitats where food may be more abundant (Haaker *et al.* 1986). Movement patterns have been further characterized in green lip abalone by Cennie *et al.* (2009a), where individuals belonged to either a sedentary or a wandering clade (which moved over larger areas and covered greater distances, occupied inner portions of hides more frequently, responded more rapidly to food odor, foraged for longer periods, and displaced conspecifics from food patches).

Life span. Black abalone lifespan is unknown, but abalone are believed to survive for 30 years or more (Blecha *et al.* 1992).

Status and trends. On January 11, 2008, NMFS published a proposed rule to list black abalone as endangered under the ESA (73 FR 1986). Species decline is reflected by the decrease in commercial catches until 1993, when commercial harvests were halted. Historic levels approached 2,200 tons in California in 1879 and declined to around 1000 tons in the 1970's. Commercial landings then decreased to 19.1 tons in the last year of harvests, when mortality from withering syndrome devastated remaining black abalone stocks throughout southern

California (Haaker 1994). Over 20 years, densities of more than 100 individuals per cubic yard disappeared from most of their former range south of Point Conception (Davis 1993). A similar mass mortality was reported at Palos Verdes Peninsula in the late 1950's, where average density decreased from more than 2.8 individuals per square yard from 1975 to 1979 down to about 0.03 individuals per square yard from 1987 to 1991 (Cox 1962). Island habitats experienced more severe trends; 99% of black abalone vanished from Anacapa, Santa Barbara, and Santa Rosa Islands in less than 5 years (Haaker *et al.* 1989, Richards and Davis 1993).

Black abalone have also experienced severe declines due to a temperature-related disease called withering syndrome. This bacteria-based disease prevents assimilation of nutrients in the digestive system and results in abalone that "wither" as individuals consume body tissues. The disease was first identified west of Santa Cruz and Anacapa islands in 1985 and 1986 before spreading to Santa Rosa Island and Santa Barbara Island by 1988. The disease made its appearance along the mainland in 1988 in San Luis Obispo county, where 85% of the resident black abalone died in Diablo Cove. This die-off was attributed to the presence of warm-water effluent from a nuclear power facility. From 1988 to the early 1990's, withering syndrome continued to spread throughout the Channel islands to 2000, when it was estimated that only 1% of the original population remained (Richards 2000).

Natural threats. Along with depletion from commercial harvests, withering syndrome has also been significant in black abalone decline. Withering syndrome is a chronic, degenerative disease responsible for mass mortalities (Moore *et al.* 2000a). Warm temperature, although not associated with the initiation of withering syndrome, is associated with increased mortality rates (Lafferty and Kuris 1993, Harvell *et al.* 2009). This has been observed from power plant effluent and incursions of warm water into traditionally temperate regions. In red abalone, higher rates of infection and more prominent signs of infection are associated with El Niño events (Moore *et al.* 2009). Interspecific competition has not been studied in black abalone, but blacklip abalone face significant competition for algal resources by sea urchins (Strain and Johnson 2009).

Compounding these factors are reproductive factors that further hamper species recovery. At low densities, individuals aggregate for spawning are not close enough for fertilization to occur. As a result, annual recruitment of juvenile black abalone has declined steeply since adult populations dropped below half of initial densities (Richards and Davis 1993).

Black abalone at various life stages experience predation from several species. Juvenile abalone hiding amongst rocks are food for crabs, lobsters, octopi, starfish, fish, and predatory snails (Haaker *et al.* 1986). Abalone of intermediate sizes are vulnerable to octopus and fish predation, particularly sheepshead and cabezon. As adults, black abalone are primarily preyed upon by sea otters, which can be major regulators of black abalone populations, but are not know to extirpate communities as other threats are known to (Braje *et al.* 2009, Johnson *et al.* 2009). Interactions with other species can hinder species recovery in other ways, namely competition for space and food resources. Purple and red sea urchins tend to feed on the same kelp and brown algae food as black abalone and, when in high abundance and food is plentiful, have the potential to out-compete abalone for food (Leighton 1968, Paine 1974, Tegner and Levin 1982, Tegner 1989, Miller and Lawrenz-Miller 1993). However, abalone tend to inhabit different habitats than these urchin species (CDFG 1993). Space competition may also occur between black abalone and sand castle worms. This species cements itself to the underside of rocks, the same habitat that black abalone seek for refuge (Connell *et al.* 1988). This could limit the habitat available for black abalone to recruit into during recovery. Other factors that can threaten black abalone include storms (crushing abalone between rocks and sedimentation in rocky habitat), fresh water input, sedimentation on gills leading to asphyxiation, and temperature impacts on reproduction and growth (Cox 1960, 1962).

Anthropogenic threats. Although commercial harvests historically lead to black abalone depletion, current harvesting is a small fraction of those levels. However, small removals are still significant in small populations.

Although toxicology of abalone is poorly known, red abalone have demonstrated metabolic breakdown when exposed to the pesticide 3-trifluoromethyl-4-nitrophenol (Viant *et al.* 2001). Silver, cadmium, and mercury are also known to bioaccumulate in abalone, likely from ingested algae (Huang *et al.* 2008). Update rates have been measured as 1.78 L g^{-1} d^{-1} for silver, 0.056 L g^{-1} d^{-1} for cadmium and 0.32 L g^{-1} d^{-1} for mercury, of which 58 to 83%, 33 to 59%, and 65 to 78%, respectively, is assimilated. Abalone are known to bioaccumulate high levels of heavy metals in the presence of high environmental concentrations (Wang *et al.* 2009).

Changes in sea surface temperatures have been suggested as a driving force in altering red abalone distribution in the past (Braje *et al.* 2009); it is unknown what affect, if any climate change may have on black abalone.

Critical habitat. On October 27, 2011, the NMFS designated critical habitat for black abalone This includes rocky areas from mean high water to six meters water depth in the Farallon, Channel, and Año Nuevo islands, as well as the California coastline from Del Mar Ecological Reserve south to Government Point (excluding some stretches, such as in Monterey Bay and between Cayucos and Montaña de Oros State Park) in northern and central California and between the Palos Verdes and Torrance border south to Los Angeles Harbor. These areas include primary constituent elements required by black abalone, such as rocky substrates to cling to, food resources (bacterial and diatom films, crustose coralline algae, and a source of detrital macroalgae), juvenile settlement habitat (rocky intertidal habitat containing crustose coralline algae and crevices or cryptic biogenic structures (e.g., urchins, mussels, chiton holes, conspecifics, anemones)), suitable water quality (temperature, salinity, pH, and other chemical characteristics necessary for normal settlement, growth, behavior, and viability of black abalone), and suitable nearshore circulation patterns (where sperm, eggs, and larvae are retained in the nearshore environment).

Marine Plants

Johnson's seagrass

Description of the species. Johnson's seagrass has only relatively recently been identified as a distinct species and therefore no historical distribution information is available (Eiseman and McMillan 1980).

Distribution. Current distribution includes lagoons along approximately 125 miles of southeastern Florida between Sebastian Inlet and north Biscayne Bay which means that Johnson's seagrass has the most limited geographic distribution of any seagrass in the world (Kenworthy 1997). However, northern range extentions (likely temporary) have recently been observed (Virnstein and Hall 2009). The largest known groups of patches are located near Sebastian Inlet and Lake Worth.

Habitat. Patches of Johnson's seagrass have been observed to grow from the intertidal zone down to 3.3 feet water depth and in waters with variable temperatures and salinities (15 to 43 parts per thousand) and temperatures (Dawes *et al.* 1989, Kenworthy 1993, Virnstein *et al.* 1997, Kahn and Durako 2009). Patches near freshwater discharges have been observed (Gallegos and Kenworthy 1996), although Torquemada et al (2005) noted that highly hypo- or hypersaline conditions can negatively impact growth. Intertidal patches may be completely exposed at low tides, suggesting tolerance to desiccation and wide temperature ranges (Kahn and Durako 2009).

Growth and reproduction. Only female flowers have been observed; no fruit or seeds have been found to date (Eiseman and McMillan 1980, Heidelbaugh *et al.* 2000). Meiosis does occur however, meaning that if male pollen ware even rarely present, sexual reproduction could take place (York 2005). However, there is no evidence of male flowers, meaning Johnson's seagrass probably reproduces by cloning or asexual branching and fragmentation (Jewitt-Smith *et al.* 1997, Hammerstom and Kenworthy 2003). Consequently, genetic diversity is low (Freshwater and York 1999), putting Johnson's seagrass at a potential genetic disadvantage compared to other seagrasses, particularly if removed from an area. However, if male pollen are even rarely present, sexual reproduction could take place (York 2005).

Clonal reproduction occurs when plants form new leaf-pair, root and rhizome segments that arise from terminal buds (Posluszny and Tomlinson 1990). On average, new buds are formed on rhizomes every two to four days and rhizomes can grow at 0.2 inch per day (Bolen 1997, Kenworthy 1997). However, these clones can expand rapidly (1 to 3 feet per month) during periods of prolific branching (Kenworthy 1997, Greening and Holland 2003, Kenworthy 2003). As clones expand, high density "patches" are formed ranging from three to 66 feet^2 in size (Kenworthy 1997, Virnstein *et al.* 1997, Kenworthy 2000, 2003, Virnstein and Morris 2007). Patches can expand rapidly (nine feet^2 per month)(Kenworthy 2003) leading to coalescence with adjacent patches and large meadows of up to 30 acres (Kenworthy 1997).

Fragments or entire plants can be uprooted and drift extensively, providing a mechanism for dispersal and colonization of new areas (Hall *et al.* 2006). Virnstein *et al.* (2009) recently proposed that Johnson's seagrass occurs in "pulsating patch," with two to three consecutive summers of growth followed by a rapid decline. Johnson's seagrass frequently undergo whole patch mortality followed by recolonization (Virnstein *et al.* 1997, Heidelbaugh *et al.* 2000, Greening and Holland 2003, Kenworthy 2003, Virnstein and Morris 2007).

Johnson's seagrass appears to be physiologically adapted to exploit unstable environments and unvegetated patches, with minimal resources allocated to the holding of space (Dean and Durako 2007). This characteristic may allow for more rapid overall patch growth and the exploitation of areas in which Johnson's seagrass could not otherwise compete (Dean and Durako 2007). These growth characteristics also help explain its patchy distribution (Kenworthy 1993, Virnstein *et al.* 1997). Although successful in unstable areas, Johnson's seagrass may be out-competed by more stable-selected plants in areas not subject to regular disturbance (Durako 2003). Johnson's seagrass thrive in unstable or newly-created unvegetated environments, but have little capacity for holding occupied space. As a result, Johnson's seagrass can be highly variable in its occurrence over relatively short time frames (Virnstein *et al.* 2009). Due to this species' physiology, low capacity for storage, and shallow root system, growth over large unsuitable patches may be unlikely, and its ability to recover from widespread habitat loss may be limited.

Status and trends. On September 14, 1998, Johnson's seagrass was listed as threatened under the ESA (69 FR 49035).

Historical abundance estimates of Johnson's seagrass are not available due to the species having only recently been differentiated. Limited data indicate no large distributional gaps or changes in abundance over much of Johnson's seagrass distribution from 1994 to 1999. However, recent increases in reported occurrence could be an artifact of recent increases in search efforts.

Natural threats. Storms pose the greatest natural threat to Johnson's seagrass. Storms can easily uproot or rip apart individuals and scatter them widely. Although this can serve to disperse individuals into new habitats, it can also catastrophically eliminate established meadows. Subsequent siltation following high turbidity events can also bury individuals or parts of plants.

Anthropogenic threats. Due to its delicate morphology, small range, lack of genetic diversity and a physiology ill equipped to hold space and compete with other seagrasses, Johnson's seagrass is vulnerable to prolonged widespread human-induced disturbance and habitat loss and its potential for recovery may be limited. Johnson's seagrass and its habitat are threatened by several natural and anthropogenic factors, including (1) dredging and filling, (2) construction and shading from in- and overwater structures, (3) prop scarring and anchor mooring, (4) trampling, (5) altered water quality (such as stormwater runoff and turbidity), and (6) siltation, as well as climate change (Waycott *et al.* 2009).

Critical habitat. Critical habitat for Johnson's seagrass was designated on April 5, 2000 (65 FR 17786) and includes (1) locations with populations that have persisted for 10 years; (2) locations with persistent flowering populations; (3) locations at the northern and southern range limits of the species; (4) locations with unique genetic diversity; and (5) locations with a documented high abundance of Johnson's seagrass compared to other areas in the species' range. These PCEs are critical to the conservation of the species because they protect persistently reproductive and genetically diverse populations, allow for protective buffers along the distribution limits (i.e., edges of survival), and protect regions of high density that without further knowledge of species biology, appear to serve the needs of Johnson's seagrass. Ten regions of sheltered bay and inlet waters are designated, including north and south of Sebastian Inlet, near Fort Pierce Inlet, north of St. Lucie Inlet, a portion of Hobe Sound, the southern side of Jupiter Inlet, Lake Worth Lagoon (north of Bingham Island and Boynton Inlet), waters of Lake Wyman, and wide areas of northern Biscayne Bay. These regions occupy approximately 22,574 acres or 9,139 hectares. Simply the nature of Johnson's seagrass critical habitat makes it variable and prone to change. The growth of boating in Florida and development of coastal areas has resulted in trampling, propeller scarring, dredging, filling, shading, and altered water quality that has degraded these areas compared to historical conditions. Although many of the factors that can negatively affect Johnson's seagrass and their habitat are generally well regulated and enforced, the species is still under threat from high development pressure and subsequent habitat degradation throughout its range.

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 proposed or listed species as well as their proposed or designated critical habitats in the action area.

Because this is a programmatic consultation, however, with a broad geographic scope that encompasses waters of the United States, this *Environmental baseline* serves a slightly different purpose. The *Environmental baseline* for this consultation focuses on the status and trends of the aquatic ecosystems in the United States and the consequences of that status for listed resources that occur in a general region. Since our action area and the *Environmental baseline* encompass a very broad spatial scale with many distinct ecosystems, wherever possible we have focused on common indicators of the biological, chemical, and physical health of the nation's aquatic environments. The *Environmental baseline* for this consultation provides the backdrop for evaluating the effects of the action on listed and proposed resources under NMFS' jurisdiction.

We divided the *Environmental baseline* for this consultation into marine versus freshwater regions. The freshwater component includes estuaries as well as five broad geographic regions: the Northeast Atlantic Region, the Southeast Atlantic Region, the Gulf Coast Region, the Southwest Region, and the Pacific Northwest Region. In some instances regions were further subdivided according to ecoregions, importance to NMFS' trust resources or other natural features. In each freshwater section we described the biological and ecological characteristics of the region such as the climate, geology, and predominant vegetation to provide landscape context and highlight some of the dominant processes that influence the biological and ecological diversity of the region where proposed, threatened, and endangered species reside. We then described the predominant land and water uses within a region to illustrate how the physical and chemical health of regional waters and the impact of human activities have contributed to current status of listed and proposed resources.

Stressors within the marine environment tend to be much more ubiquitous than in freshwater ecosystems and thus we have not generally divided stressors in the marine environment into more specific components, although some areas are relatively unique in regards to some stressors, such as oil and gas industrial activities or hurricane impacts, and are described in a more regional context.

Climate change

We primarily discuss climate change as a threat common to all species addressed in this Opinion, rather than in each of the species-specific narratives. As we better understand responses to climate change, we will address these effects in relevant species-specific sections.
In general, based on forecasts made by the Intergovernmental Panel on Climate Change (IPCC), climate change is projected to have substantial direct and indirect effects on individuals, populations, species, and the structure and function of marine, coastal, and terrestrial ecosystems in the near future (IPCC 2000, 2001b, a, 2002). From 1906-2006, global surface temperatures have risen 0.74º C and continues at an accelerating pace; 11 or the 12 warmest years on record since 1850 have occurred since 1995 (Poloczanska *et al.* 2009). Furthermore, the Northern Hemisphere (where a greater proportion of ESA-listed species occur) is warming faster than the Southern Hemisphere, although land temperatures are rising more rapidly than over the oceans (Poloczanska *et al.* 2009). The direct effects of climate change will result in increases in atmospheric temperatures, changes in sea surface temperatures, patterns of precipitation, and sea level. 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. Species that are shorter-lived, of larger body size, or generalist in nature are liable to be better able to adapt to climate change over the long term versus those that are longer-lived, smaller-sized, or rely upon specialized habitats (Purvis *et al.* 2000, Brashares 2003, Cardillo 2003, Cardillo *et al.* 2005, Issac 2009). Climate change is most likely to have its most pronounced affects 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.

Some indirect effects of climate change would result from changes in the distribution of temperatures suitable for whale calving and rearing, the distribution and abundance of prey and abundance of competitors or predators. For species that undergo long migrations, individual movements are usually associated with prey availability or habitat suitability. If either is disrupted, 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 low-success of northern elephant seals during El Niño periods, when cooler, more productive waters are associated with higher first year pup survival (McMahon and Burton. 2005). Reduced prey availability resulting from increased sea surface temperatures has also been suggested to explain reductions in Antarctic fur seal pup and harbor porpoise survival (Forcada *et al.* 2005, Macleod *et al.* 2007). Polygamous marine mammal mating systems can also be 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). For marine mammals considered in this Opinion, available data suggest sperm whale females have lower rates of conception following periods of unusually warm sea surface temperature (Whitehead 1997). Marine mammals with restricted distributions linked to water temperature may be particularly exposed to range restriction (Learmonth *et al.* 2006, Issac 2009). MacLeod (2009) estimated that, based upon expected shifts in water temperature, 88% of cetaceans would be affected by climate change, 47% would be negatively affected, and 21% would be put at risk of extinction. Of greatest concern are cetaceans with ranges limited to non-tropical waters and preferences for shelf habitats (Macleod 2009). 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 coverage 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 1841-1995 and concluded Antarctic sea ice cover had declined by about 20% since the 1950s.

Roughly 50% of the Earth's marine mammal biomass occurs in the Southern Ocean, with all baleen whales feeding largely on a single krill species, *Euphausia superba*, here and feeding virtually nowhere else (Boyd 2002). Atkinson *et al.* (2004) linked sea ice loss to severe decreases in krill populations over the past several decades in some areas of the Antarctic. Reid and Croxall (2001) analyzed a 23-year time series of the reproductive performance of predators (Antarctic fur seals, gentoo penguins, macaroni penguins, and black-browed albatrosses) that depend on krill for prey and concluded that these populations experienced increases in the 1980s followed by significant declines in the 1990s; overall an increase in the frequency of years with reduced reproductive success occurred. These declines resulted, at least in part, from changes in the structure of the krill population, particularly reduced recruitment into older krill age classes, which lowered the number of predators krill could sustain. The authors concluded that the biomass of krill within the largest size class was sufficient to support predator demand in the 1980s but not in the 1990s. By 2055, severe reductions in fisheries catch due to climate change have been suggested to occur in the Indo-Pacific, Red Sea, Mediterranean Sea, Antarctic, and tropical areas worldwide while increased catches are expected in the Arctic, North Pacific, North Atlantic, and northern portions of the Southern Ocean (Cheung *et al.* 2010).

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

Climate-mediated changes in the distribution and abundance of keystone prey species like krill and climate-mediated changes in the distribution of cephalopod populations worldwide is likely to affect marine mammal populations as they re-distribute throughout the world's oceans in search of prey. If sea ice extent decreases, then larval krill may not be able to survive without access to underice algae to feed on. This may be a cause of decreased krill abundance in the northern western Antarctic Peninsula during the last decade (Fraser and Hofmann 2003). Meltwaters have also reduced surface water salinities, shifting primary production along the Antarctic Peninsula (Moline *et al.* 2004). Blue whales, as predators that specialize in eating krill, are likely to change their distribution in response to changes in the distribution of krill (Payne *et al.* 1986, Payne *et al.* 1990, Clapham *et al.* 1999). If they did not change their distribution or could not find the biomass of krill necessary to sustain their population numbers, their populations (and other large whales with similar life histories, such as humpback whales) would likely experience declines similar to those observed in other krill predators, including dramatic declines in population size and increased year-to year variation in population size and demographics. These outcomes would dramatically increase the extinction probability of baleen whales. Edwards *et al.* (2007) found a 70% decrease in one zooplankton species in the North

Sea and an overall reduction in plankton biomass as warm-water species invade formerly cold-water areas. Productivity may increase in other areas, though, providing more resources for local species (Brown *et al.* 2009). In addition, reductions in sea ice may alleviate "choke points" that allow some marine mammals to exploit additional habitats (Higdon and Ferguson 2009).

The indirect effects of climate change would result from changes in the distribution of temperatures suitable for reproduction, 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). For sea turtles, warming ocean temperatures may extend poleward the habitat which they can utilize (Poloczanska *et al.* 2009). Seagrass habitats have declined by 29% in the last 130 years and 19% of coral reefs have been lost due to human degradation, reducing lower latitude habitat for some sea turtle species (Poloczanska *et al.* 2009). Primary production is estimated to have declined by 6% between the early 1980s and 2010, making foraging more difficult for marine species (Hoegh-Guldberg and Bruno 2010).

Foraging is not the only potential aspect that climate change could influence. Acevedo-Whitehouse and Duffus (2009) proposed that the rapidity of environmental changes, such as those resulting from global warming, can harm immunocompetence and reproductive parameters in wildlife to the detriment of population viability and persistence. An example of this is the altered sex ratios observed in sea turtle populations worldwide (Mazaris *et al.* 2008, Reina *et al.* 2008, Robinson *et al.* 2008, Fuentes *et al.* 2009a). This does not yet appear to have affected population viabilities through reduced reproductive success, although average nesting and emergence dates have changed over the past several decades by days to weeks in some locations (Poloczanska *et al.* 2009). However, such a fundamental shift in population demographics causes a fundamental instability in population viability. Altered ranges can also result in the spread of novel diseases to new areas via shifts in host ranges (Simmonds and Eliott. 2009). It has also been suggested that increases in harmful algal blooms could be a result from increases in sea surface temperature (Simmonds and Eliott. 2009).

Changes in global climatic patterns will likely have profound effects on the coastlines of every continent by increasing sea levels and the intensity, if not the frequency, of hurricanes and tropical storms (Wilkinson and Souter 2008). A half degree Celsius increase in temperatures during hurricane season from 1965 to 2005 correlated with a 40% increase in cyclone activity in the Atlantic. Sea levels have risen an average of 1.7 mm/year over the 20th century 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 (Wilkinson and Souter 2008, Arndt *et al.* 2010, Hoegh-Guldberg and Bruno 2010). Based on computer models, these phenomena would inundate nesting beaches of sea turtles, change patterns of coastal erosion and sand accretion that are necessary to maintain those beaches, and would increase the number of turtle nests destroyed by tropical storms and hurricanes (Wilkinson and Souter 2008), although other areas might experience less frequent tropical activity and a subsequent reduction in tropical cyclone impacts to sea turtle nests (Fuentes and Abbs 2010). The loss of nesting beaches, by itself, would have catastrophic effects on sea turtle populations globally if they are unable to colonize new beaches that form or if the beaches do not provide the habitat attributes (sand depth, temperatures regimes, and refuge) necessary for egg survival. In some areas, increases in sea level alone may be sufficient to inundate sea turtle nests and reduce hatching success (Caut *et al.* 2009a). Storms may also cause direct harm to sea turtles, causing "mass" strandings and mortality (Poloczanska *et al.* 2009). Increasing temperatures in sea turtle nests alters sex ratios, reduces incubation times (producing smaller hatchlings), and reduces nesting success due to exceeded thermal tolerances (Fuentes *et al.* 2009b, Fuentes *et al.* 2009c, Fuentes *et al.* 2010). Smaller individuals likely experience increased predation (Fuentes *et al.* 2009b). Taken together, the body of literature on climate change supports widespread and significant negative consequences to sea turtle species.

Climate change also poses significant hazards to the survival and recovery of salmonids. Ongoing global climate change has implications for the current and likely future status of salmon, but particularly so in the Pacific northwest, where snow melt into the Columbia River Basin has significant influence on regional hydrology. Recent studies, particularly by the Independent Scientific Advisory Board (ISAB), describe the potential impacts of climate change in the Columbia River Basin. These effects may decrease snowfall, increase early-year runoff, decrease summer and fall flow, and generally increase water temperatures. The ISAB (2007a) identified the following list of likely effects of projected climate changes on Columbia Basin salmon:

1. Water temperature increase resulting in loss of cold-water habitat (temperatures exceed upper thermal limits for a species). Projected salmon habitat loss would be most severe in Oregon and Idaho, possibly higher than 40% of 2007 by 2090. Habitat loss would be less extreme in Washington at 22% by 2090. However, this assumes a high rate of greenhouse gas emissions and used a climate model that projected a 5º C in global temperatures by 2090, a value that is higher than the scenarios considered most likely (ISAB 2007a). Although a liberal estimate of change, this does not account for changes to hydrology that could further imbalance salmon habitat.
2. Variations in rainfall intensity may alter seasonal hydrography. With reduced snowpack and greater rainfall, the timing of stream flow will likely change, reducing spring and summer stream flow and increasing peak river flows (ISAB 2007a). This reduction in stream flow may impact the quality and quantity of tributary rearing habitat, greatly affecting spring and summer salmon and steelhead runs. In addition, the Pacific northwest's low late-summer and early-fall stream flows are likely to be further reduced, which will limit juvenile fall Chinook and chum salmon shallow mainstem rearing habitat.
3. Considering both the water temperature and hydrologic effects of climate change, abundance Snake River spring/summer Chinook populations would be substantially decreased (20-50% decline from simulated average abundance based on historical 1915-2002 climate; (Crozier *et al.* 2008). This significantly increases extinction risks in the long term.
4. Eggs of fall and winter spawning fish, including Chinook, Coho, chum, and sockeye salmon, may suffer higher levels of mortality when exposed to increased flood flows.
5. Increases in seasonal mainstem Snake and Columbia River water temperature would accelerate the rate of egg development of fall Chinook that spawn in the mainstem of

the Snake and Columbia rivers and lead to earlier (smaller size) hatching. Potential effects of increased water temperatures on adult salmon include delay in dam passage, failure to enter fish ladders, increased fallback, and loss of energy reserves due to elevated metabolic demand. Thermal stress may also lead to increased risk of parasitism and disease.

6. Earlier snowmelt and higher spring flows, warmer temperatures, more rain, and less snow may cause spring Chinook and steelhead yearlings to smolt and emigrate to the estuary and ocean earlier in spring. The early emigration coupled with a projected delay in the onset of coastal upwelling could cause these fish to enter the ocean before foraging conditions are optimal. The first few weeks in the ocean are thought to be critical to the survival of salmon off Oregon and Washington, so a growing mismatch between smolt migrations and coastal upwelling would likely have significant negative impacts on early ocean survival rates.
7. Within the Columbia estuary, increased sea levels in conjunction with higher winter river flows could degrade estuary habitats. Numerous warm-adapted fish species, including several non-indigenous species, normally found in freshwater have been reported from the estuary and might expand their populations with the warmer water. Climate change also may affect the trophic dynamics of the estuary due to upstream extension of the salt wedge in spring-early summer caused by reduced river flows. Changes in the upstream extension of the salt wedge will influence the location of fish prey, but it is difficult to forecast the effect this change will have on juvenile salmon.
8. Physical changes in the ocean associated with warming include increases in temperature, increased water column stratification, and changes in the intensity and timing of coastal upwelling. These changes will alter primary and secondary productivity, the structure of marine communities, and, in turn, the growth, productivity, survival, and migrations of salmonids.
9. Changing ocean temperatures may alter salmon behavior, distribution, and migrations, increasing the distance from home streams to ocean feeding areas. Energetic demands increase at warmer temperatures, requiring increased feeding to maintain growth. This could lead to intensified competition for food and reduction in growth rates, further exacerbating the prey/predator relationship.
10. Increasing concentrations of carbon dioxide in the oceans lowers pH, which reduces the availability of carbonate for shell-forming marine animals. Pteropods are expected to be negatively affected, and they can comprise more than 40% of some salmon diets. If salmon migrate farther to the north and/or food is less available, longer times may be required to reach maturity, delaying the usual times of adult migrations into coastal water and rivers.

Climactic shifts also occur due to natural phenomenon. In the North Atlantic, this primarily concerns fluctuations in the North Atlantic Oscillation (NAO), which results from changes in atmospheric pressure between a semi-permanent high pressure feature over the Azores and a subpolar low pressure area over Iceland (Hurrell 1995, Curry and McCartney 2001, Stenseth *et al.* 2002). This interaction affects sea surface temperatures, wind patterns, and oceanic circulation in the North Atlantic (Stenseth *et al.* 2002). The NAO shifts between positive and negative phases, with a positive phase having persisted since 1970 (Hurrell 1995). North Atlantic conditions experienced during positive NAO phases include warmer than average winter weather in central and eastern North America and Europe and colder than average temperatures Greenland and the Mediterranean Sea (Visbeck 2002). Effects are most pronounced during winter (Taylor *et al.* 1998). The NAO is significant for North Atlantic right whales due to its influence on the species primary prey, zooplankton of the genus *Calanus*, which are more abundant in the Gulf of Maine during positive NAO years (Conversi *et al.* 2001, Drinkwater *et al.* 2003, Greene *et al.* 2003a, Greene and Pershing 2004, Kiszka *et al.* 2010a). This subsequently impacts the nutritional state of North Atlantic right whales and the rate at which sexually mature females can produce calves (Greene *et al.* 2003a). Local distribution shifts of North Atlantic right whales may be tied to the NAO (Kenney 2007).

Periodic weather patterns such as El Niño, La Niña, and the Pacific decadal oscillation (PDO) can fundamentally change oceanographic conditions in the northeastern Pacific and the biology that is based upon it (Stabeno *et al.* 2004, Mundy and Cooney 2005, Mundy and Olsson 2005). Roughly every 3-7 years, El Niño can influence the northeastern Pacific (JOI/USSSP 2003, Stabeno *et al.* 2004). Typical changes include increased winter air temperature, precipitation, sea level, and downwelling favorable conditions (Royer and Weingartner 1999, Whitney *et al.* 1999). La Niña events tend to swing these conditions in the negative direction (Stabeno *et al.* 2004). However, sea surface temperatures (SSTs) can take 1 year to change following an El Niño event or change to varying degrees (Freeland 1990, Bailey *et al.* 1995, Brodeur *et al.* 1996a, Royer 2005). Eddies occur with greater frequency in the Gulf of Alaska during El Niño years; the corollary is true during La Niña years (Crawford *et al.* 1999, Melsom *et al.* 1999). The 1997/1998 El Niño event is attributed with reducing primary production in the Gulf of Alaska by at least 50% (likely due to nitrate depletion); zooplankton abundance decreases as their phytoplankton food supply diminishes. Phytoplankton in continental shelf waters tends not to be effected by these events (Freeland 2000, Coyle and Pinchuk 2003, Zamon and Welch 2005b). The 1982/1983 El Niño and other downwelling events are generally regarded to have reduced food supplies for marine mammals along the US west coast (Feldkamp *et al.* 1991, Hayward 2000, Le Boeuf and Crocker 2005). Marine mammal distribution and social organization (group size) is also believed to have shifted northward in response to persistent or extralimital prey occurrence in more northerly waters during El Niño events (Shane 1994, 1995, Benson *et al.* 2002, Lusseau *et al.* 2004, Norman *et al.* 2004, Danil and Chivers 2005). Low reproductive success and body condition in humpback whales have also been suggested to have resulted from the 1997/1998 El Niño (Cerchio *et al.* 2005). El Niño events in the winters of 1952-1953, 1957-1958, 1965-1966, and 1982-1983 were associated with strong downwelling anomalies, which reduces nutrient availability for plankton (Bailey *et al.* 1995, Wheeler and Hill 1999, Thomas and Strub 2001). Plankton diversity also shifts, as smaller plankton are better able to cope with reduced nutrient availability (Corwith and Wheeler 2002, Sherr *et al.* 2005).

The PDO is the leading mode of variability in the North Pacific and operates over longer periods than either El Niño or La Niña and is capable of altering SST, surface winds, and sea level pressure (Mantua 2002, Mantua and Hare 2002, Stabeno *et al.* 2004). Unlike El Niño and La Niña events, PDO events can persist for 20-30 years, are more prominent outside the tropics, and mechanisms controlling them are relatively unknown (Minobe 1997, 1999, Hare and Mantua 2000, Mantua and Hare 2002). During positive PDOs, the northeastern Pacific experiences above-average SSTs while the central and western Pacific Ocean undergoes below-normal SSTs (Mundy and Olsson 2005, Royer 2005). Warm PDO regimes, as with El Niño events, tends to decrease productivity along the US west coast (Hare *et al.* 1999, Childers *et al.* 2005). However, during the 1977 warm phase of the PDO, euphausiid biomass remained the same and copepod abundance actually increased in the Pacific northwest; zooplankton biomass doubled in offshore waters of the Gulf of Alaska (Brodeur and Ware 1992, McFarlane and Beamish 1992, Brodeur *et al.* 1996b, Francis and Hare 1997, MacCall *et al.* 2005). Opposite SST regimes occur during negative PDOs (Mundy and Olsson 2005). Positive PDOs occurred from 1925-1946 and 1977-1999. Negative PDOs occurred from 1890-1924, 1947-1976, and 1999-present (Mantua *et al.* 1997, Minobe 1997, Childers *et al.* 2005).

The potential for invasive species to spread under the influence of climactic change is also a significant concern. If water temperatures warm in marine ecosystems, native species may shift poleward to cooler habitats, opening ecological niches that can be occupied by invasive species introduced via ships ballast water or other sources (Ruiz *et al.* 1999, Philippart *et al.* 2011). A similar observation of "Caribbean creep" has been observed, with warmer waters facilitating the range expansion of warmer-water species into more northerly regions (Canning-Clode *et al.* 2011). Although these expansions may be temporary, they can include harmful algal bloom species whose presence even temporarily can cause major morbidity and mortality issues to a variety of endemic species (Hallegraeff 2010). Moore *et al.* (2011) estimated that the impacts of a dinoflagellate establishment would likely intensify with a warming climate, resulting in roughly 13 more days of potential bloom conditions per year by the end of the 21st century. Invasive species that are better adapted to warmer water temperatures can also outcompete native species that are physiologically geared towards lower water temperatures; such a situation currently occurs along central and northern California, where the Mediterranean blue mussel has established and is displacing a native mussel competitor (Lockwood and Somero 2011).

Freshwater and Coastal Estuarine Systems

Ballast Water Bioregion NA-ET2 (Maine to Massechusetts)

This region encompasses Maine, New Hampshire, and Massachusetts. The region is ecologically diverse, encompassing several broad ecoregions—according to Bailey's (1995) *Description of the Ecoregions of the United States* this region encompasses the warm continental, the hot continental and the hot continental mountains divisions —these ecoregions can be further subdivided into provinces based on vegetation (Bailey 1995). This region encompasses the New England/Acadian mixed forests and the Northeastern Coastal Forests.

In this section, we describe several basins and estuarine complexes to characterize the general ecology and natural history of the area, and past and current human activities and their impacts on the area. In certain instances we described some river basins in further detail to provide additional context for evaluating the influence of the environmental baseline on listed species under NMFS' jurisdiction and the health of the environment.

Natural History. This region encompasses drainages entering the Gulf of Maine, and encompasses all of Maine, parts of New Hampshire, Massachusetts the Canadian provinces of

New Brunswick and Nova Scotia. Characterized by a temperate climate and a rocky coastline, the greater Gulf of Maine encompasses the Bay of Fundy, Casco Bay, Massachusetts Bay, Merrymeeting Bay and Cape Cod Bay. Significant Rivers that drain into the Gulf of Maine include the St. John, St. Croix, Penobscot River Basin, Kennebec/Androscoggin River Basin and the Merrimack River Basin.

Estuaries within the Gulf of Maine were formed by glaciers and as a result have characteristically rocky shorelines, shallow soils, and deeply carved channels. The Gulf of Maine is semi-enclosed—bounded to the south by Georges Banks and to the north by Brown's Bank. The area is more strongly influenced by the Labrador Current, which makes the waters significantly colder and more nutrient rich than waters to the south that are more strongly influenced by the Gulf Stream.

The cold waters of the Gulf of Maine make it one of the most productive marine ecosystems in the world. The Gulf is characterized by salt marshes, kelp and seagrass beds, tidal mudflats, and underwater rocky outcrops form the foundation of a complex ecosystem and provide habitat for Atlantic herring (*Clupea harengus*), American lobster (*Homarus americanus*), Atlantic salmon, several whale species including endangered Northern right whales—where they are regularly observed in the spring and summer at regular nursery and feeding areas.

Penobscot River Basin. The Penobscot River flows 275 miles to the ocean, with the largest watershed in Maine of 8,592 square miles (mi^2) (Jackson *et al.* 2005). The river flows from the mountains of western Maine, including Maine's highest peak, Mt. Katahdin to the ocean near the town of Bucksport, Maine. The Penobscot basin was formed by glaciation during the last ice age and the river's bed is composed of glacial deposits and granitic bedrock. The average precipitation is approximately 42 inches per year. At the mouth, the average discharge is 10.1 billion gallons each day, or 14,000 cubic feet per second, but the discharge fluctuates seasonally and with dam releases, with naturally higher flows in the spring (Hasbrouck 1995(MaineRivers 2007). The river and estuary are also important for many fish species, with 45 freshwater and 39 salt water species having been recorded in the river or estuary. Despite being home to so many fish, there are only three nonnative species (Baum 1983(Jackson *et al.* 2005). The Penobscot estuary extends from Bangor downstream to Penobscot Bay in the Gulf of Maine, approximately 31 miles, making it the largest estuary in Maine and one of the largest on the East Coast (PEARL 2007). Downstream of Bangor, the river is a tidally influenced, salt-wedge estuary. The majority of the estuary is bedrock-based, and sediment deposits are limited to isolated coves and near marshes.

Merrymeeting Bay Basin. Merrymeeting Bay is the largest, freshwater tidal estuary, approximately 18.6 miles upstream of the mouth of the estuary that enters the Gulf of Maine (Kistner and Pettigrew 2001, Jackson *et al.* 2005). The Kennebec and Androscoggin Rivers, along with four smaller tributaries, converge to form the bay, although the two large rivers account for 98% of the inflow. Merrymeeting Bay typically has the largest freshwater outflow to the Gulf of Maine, usually exceeding 15,000 cubic feet per second. These high flows thoroughly flush the bay and have prevented eutrophication. The bay substrate is mud, sand, and exposed bedrock.

In Merrymeeting Bay, sampling only sandy substrate, which doesn't hold as much contaminant as muddy substrates due to less surface area, some toxic substances were identified. Sediments associated with the Androscoggin River had higher levels of PAHs and mercury, while sediments from the Kennebec River had higher levels of chromium, arsenic, and selenium (Hayden 1998). The bay has more moderate levels of these toxins than the rivers themselves. Chilcote and Waterfield (1995) found that levels of arsenic are higher than levels identified by EPA as likely to have adverse effects. At one station, PAHs from the Androscoggin also exceeded EPA identified levels of minimal effects. In this region of the Gulf of Maine, metal deposition is linked more to the Androscoggin and Kennebec than the Sheepscot River. Based on benthic samples taken in 1980 and again in 1991, it appears that all metals are declining in Merrymeeting bay except for copper, which showed an increase (Hayden 1998). Commercially important fish also have elevated metal concentrations in their livers, which is thought to be from their time spent in Merrymeeting Bay (Kistner and Pettigrew 2001).

The Kennebec River flows 230 miles from the headwaters to the ocean, with a watershed of 5,384 mi^2 (Jackson *et al.* 2005, Maine Rivers 2007). The Kennebec River basin is primarily medium to coarse sand with some glacial till overlaying bedrock. Average precipitation is 42.5 inches of rain per year (Jackson *et al.* 2005). The average discharge at the mouth of the Kennebec River is 5,893 million gallons per day, with natural and controlled discharges similar to those seen on other Maine rivers (Maine Rivers 2007). There are 48 species of freshwater fish that use the Kennebec, including 10 nonnative species.

The Androscoggin River travels 164 miles, with a watershed of 3,263 mi^2 (Jackson *et al.* 2005)MaineRivers 2007c). The river flows from northwest Maine, into New Hampshire, and then back into Maine, where it meets the Kennebec River in Merrymeeting Bay. The Androscoggin has been Maine's principle industrial river (MaineRivers 2007c). The average precipitation in the watershed is 43.7 inches per year, resulting in an average discharge at the mouth of the Androscoggin, entering Merrymeeting Bay, of approximately 4,190 million gallons each day. The river is home to 33 freshwater fish and 7 estuarine fish, including 8 nonnative species (Jackson *et al.* 2005).

Merrimack River Basin. The Merrimack River is 180 miles long, with 16 sub-basins in a watershed of 5,014 mi^2 (Jackson *et al.* 2005, MRWCI 2007). Seventy five percent of the watershed is in New Hampshire, with the rest in northeast Massachusetts. The precipitation is approximately 36 inches per year, with an average discharge of 5,364 million gallons per day, or 8,299 cubic feet per second. The geology of the Merrimack is dominated by granitic bedrock. The river is home to 50 species of fish, including 5 nonnative species (Jackson *et al.* 2005). For the lowest nine miles of the Merrimack River, extending north into New Hampshire and south to Cape Ann, Massachusetts, there are 25,000 acres of estuarine habitat and 15,000 acres of salt marsh habitat, which is referred to as the Great Marsh (USGS 2003).

Human Activities and Their Impacts

Land Use. Most of the watersheds within this region are heavily forested with relatively small areas of highly urbanized lands. Land use in the Penobscot watershed is 5% agriculture and 95% forest and wetland (90% forest and forested wetlands). There are approximately 21 people per square mile living in the Penobscot watershed, and the largest town is Bangor, consisting of

33,000 people (Jackson *et al.* 2005). While there is not much urban development in the watershed, Doggett and Sowles (1989) report tanneries, metal finishing, pulp and paper mills, textile plants, chemical products, and municipal sewage contribute chromium, mercury, zinc, copper, lead, arsenic, hydrocarbons, dioxins, PAHs, pesticides, and other contaminants to the river.

The Kennebec River watershed usage is 82% forest, 10% water, 6% agriculture, 2% developed (Jackson *et al.* 2005). The only major town in the watershed is Augusta, Maine, but there are approximately 39 people per square mile throughout the watershed (Jackson *et al.* 2005). Currently, the primary pollution source on the river is from two pulp and paper mills, but there were multiple historical polluters along the river. The river exceeds recommended levels of dioxins, arsenic, cadmium chromium, copper, lead, mercury, nickel, silver, zinc, and PAHs in the sediments and surface water (MDEP 1999) (Harding Lawson Associates 1999, 2000). Since 1990, the levels of dioxins in other rivers in Maine have been decreasing, but the levels in the Kennebec have remained constant (Kahl 2001).

The Androscoggin River watershed usage is 5% agriculture, 86% forested, 7% water, and 2% developed (Jackson *et al.* 2005). Major towns in the Androscoggin watershed are Auburn, Lewiston, and Brunswick. The human population in the watershed is approximately 65 people per square mile (Jackson *et al.* 2005). Throughout the 20th century, textile mills, paper and pulp mills, and municipalities contributed large quantities of pollutants to the river. At one time it was considered one of the 10 most polluted rivers in the country and was one of the reasons for the implementation of the Clean Water Act. The river has become much cleaner since the Clean Water Act (CWA) was passed, but pesticides, mercury, lead, sedimentation, total suspended solids, PCBs, and dioxins are still considered too high (Chamberland *et al.* 2002).

The Merrimack River watershed is composed of 75% forest, 13% urban, 6% agriculture, 5% surface water, and 1% other (Jackson *et al.* 2005). The Merrimack River flows through industrial centers Manchester and Concord, New Hampshire, and Lowell and Lawrence, Massachusetts. There are approximately 404 people per square mile in the Merrimack watershed (Jackson *et al.* 2005). The biggest sources of pollution facing the river are combined sewage overflows, industrial discharge, urbanization and its associated run-off (USACE 2003). The upper mainstem of the river has problems with bacteria, E. coli, and acidity, while the lower mainstem has problems with bacteria, metals, nutrients, dioxins, turbidity and suspended solids, and un-ionized ammonia. In all, over 125 miles of mostly lower watershed areas do not support their designated uses (USACE 2003).

Hydromodification Projects. There are five major hydroelectric dams along the mainstem of the Penobscot River as well as 111 other licensed dams located along the river and its tributaries. Atlantic salmon historically migrated as far as 143 miles upstream of the mouth, but due to development along the river, in the 1960s, Atlantic salmon were extirpated (Jackson *et al.* 2005). The population has since been re-established and runs of 2,000 to 4,000 occur with natural spawning as far upstream as 62 miles. Unfortunately, 6,000 to 10,000 salmon are required for a sustainable population, so the Penobscot run depends on fish from a local hatchery (Moore and Platt 1997).

The Kennebec River has eight large hydroelectric dams on its mainstem, which restricts fish passage both up and downstream. In 1999, the Edwards Dam was removed, opening 17 additional miles of habitat for fish and macroinvertebrates in the river. Removal of Edwards dam restored full access to historical spawning habitat for species like Atlantic sturgeon, shortnose sturgeon, and rainbow smelt, but not for species like alewife, American shad, and Atlantic salmon that migrated much further up the river. Since the removal of Edwards Dam, DO levels and macroinvertebrate density have improved. Additionally, in 2007, the fish passage facilities on the lowest dam on the Kennebec River as well as the second and third lowest dams on the Sebasticook River became operational. The lowest dam on the Sebasticook River has been decommissioned and may be breached in as early as 2007 (MDMR 2007).

The Androscoggin River has 14 hydroelectric dams on the mainstem of the river and 18 in the watershed. Fish ladders have been installed on the lower dams allowing anadromous fish passage to Lewiston Falls (Brown *et al.* 2006). The dams play a considerable role in the poor water quality of the river, causing reduced DO throughout the summer. During the 60s, most of the river had oxygen levels of 0ppm, resulting in massive fish kills. There is still a 14 mile stretch of river that requires aerators to provide dissolved oxygen to the river.

The Merrimack River watershed has over 500 dams, including three in Massachusetts and three in New Hampshire, that essentially make the mainstem into a series of ponds (Dunn Jr 2002, Jackson *et al.* 2005). Flow alteration is considered a problem on the upper mainstem of the river and has resulted in the river not meeting EPA’s flow requirements (USACE 2003).

Mining. Mining in Northeast Atlantic watersheds first began prior to the Civil War. Since then, mining has been conducted for granite, peat, roofing slate, iron ore, sulfur, magnetite, manganese, copper, zinc, mica, and other materials. Currently, exploration for precious metals and basic metals is ongoing, but to a lesser extant than during the 1980s. Recent mining activities were conducted in this region by The Penobscot Nation, Champion Paper Company, Oquossoc Minerals, Boliden Resources, Inc., Black Hawk Mining, and BHP-Utah. There are several abandoned mines in the Northeast Atlantic coast watersheds that have become superfund sites due to excessive pollutants being leached into groundwater, such as Elizabeth Mine, Pike Hill Mine, Calhoun Mines, and others. Common pollutants leaked by mining operations in this area are lead, mercury, arsenic, and selenium (Ayuso *et al.* 2006, Piatak *et al.* 2006). All mines that are not in use are supposed to be decommissioned and cleaned up, but the impacts could persist for years before the rivers return to their pristine state.

Commercial and Recreational Fishing. The primary commercial fisheries along the Northeast Atlantic coast by harvest weight exist for herring (39%), lobster (26%), blue mussel (6%), hatchery-origin sea-run Atlantic salmon (4%), groundfish (4%), quahog (4%), soft clam (3%), sea cucumber (3%), seaweed (3%), crabs (2%), and various other species (6%). Directed harvest of shortnose sturgeon and wild Atlantic salmon is prohibited by the ESA; however, both are taken incidentally in other fisheries along the east coast and are probably targeted by poachers throughout their range (Dadswell 1979, Dovel *et al.* 1992, Collins *et al.* 1996). Since 2006, a 30 day recreational fishing season between mid September and mid October for hatchery-origin Atlantic salmon has been permitted on the Penobscot River, the only river with listed Atlantic salmon that allows salmon fishing. On the Penobscot, spring salmon fishing has not taken place since 1999, but may be permitted again in 2008. Poaching is likely another fishing threat, but its impacts to individual population segments is unknown. Entanglement of marine mammals in fishing gear is not uncommon and can lead to mortality or serious injury.

Ballast Water Bioregion NA-ET3 (Connecticut to Virginia)

This region consists of Connecticut, Rhode Island, New York, New Jersey, Delaware, Pennsylvania, Maryland and Virginia. The headwaters of the Connecticut River originate in New England/Acadian forests, and as the river descends, it transitions from boreal forest to temperate deciduous forest. As the river flows through the low gradient coastal region, the ecoregion transitions to Northeastern Coastal Forest. The headwaters of the Hudson River flow through Eastern Forest/Boreal Transition ecoregions. As the river descends, it transitions to Eastern Great Lakes Lowland Forest and then Northeastern Coastal Forest. The headwaters of the Delaware River originate in the Allegheny Highland Forest ecoregion, and then as the river descends, it transitions to Appalachian/Blue Ridge Forest and then Northeastern Coastal Forest ecoregions.

Connecticut River/Long Island Sound

Natural History. The Long Island Sound watershed includes portions of Connecticut, New York, Massachusetts, New Hampshire, Rhode Island and Vermont. Long Island Sound was designated a national estuary in 1987, due to its significance as an area where freshwater from the Connecticut, Thames, and Housatonic Rivers (90% of the freshwater input) mixes with the Atlantic Ocean. The sound ranges in salinity from 23 parts per thousand (ppt) in the western end to 35ppt on the eastern side. The surface area of Long Island Sound is 1,320 mi^2, draining an area of over 16,000 mi^2. Long Island Sound connects to the Atlantic Ocean on both the eastern and western side, called “The Race” and the East River, respectively. The sound substrate is primarily mud, sand, silt, and clay, with very small areas of exposed bedrock. The sound is home to more 120 species of fish and at least 50 species use the sound as spawning grounds.

The Connecticut River drains a watershed of 11,259 mi^2 and flows approximately 410 miles to Long Island Sound. The river flows from the highlands of New Hampshire and Quebec, and is bordered by the Green and White Mountains. The Connecticut River’s bed is composed of glacial deposits and granitic bedrock. The average precipitation is approximately 43 inches per year. At the mouth, the average discharge is 10.2 billion gallons each day, or 15,715 cubic feet per second, which accounts for approximately 70% of the freshwater inflow to Long Island Sound (Jackson *et al.* 2005). The final 56 miles of the river prior to Long Island Sound is a tidal estuary (Jackson *et al.* 2005). The river and estuary are also important for many fish species, with 64 freshwater and 44 estuarine species having been recorded in the river or estuary, but 20 of the fish are nonnative (Jackson *et al.* 2005).

Human Activities and Their Impacts

Land Use. More than eight million people live in the Long Island Sound watershed. With so many people in the watershed, both point and non-point source pollution is a major concern. Toxic substances often adsorb to the surface of sediments, which means sediments with high surface to volume ratios like sand, silt, and clay, can hold more pollutants than larger substrates. The sound has elevated levels of PCBs, PAHs, nitrogen, lead, mercury, cadmium, cesium, zinc, copper, and arsenic. Organic and metal contaminants in Long Island Sound are above national averages (Turgeon and O'Connor 1991). Lead, copper, and zinc are believed to be deposited via the atmosphere (Cochran *et al.* 1998). Cadmium, chlordane, and lead appear to be decreasing while copper is increasing (Turgeon and O'Connor 1991). Studies on winter flounder showed PAHs and PCBs leading to alteration of DNA in the livers of those fish (Gronlund *et al.* 1991). One of the biggest problems facing the sound is DO depletion (Parker and O'Reilly 1991), resulting in dead zones. The governors of Connecticut and New York have signed agreements to reduce the total nitrogen input to Long Island Sound by 58.5% before 2015 in an effort to get the DO of surface water above 5ppm, of deeper water above 3.5ppm, and no water ever below 2ppm.

Within the Connecticut River watershed the dominant land use is forest (80%), with 11% used for agriculture and the remaining 9% in mixed (other) uses (Jackson *et al.* 2005). Major towns in the Connecticut watershed are Holyoke and Springfield, Massachusetts and Hartford, Connecticut. The human population in the watershed is approximately 179 people per square mile (Jackson *et al.* 2005). Throughout the 20th century, power plants, defense contractors, municipalities, and corporations such as General Electric, Union Carbide, and Pfizer contributed large quantities of pollutants to the river. Still to this day, approximately one billion gallons of raw sewage enters the river as a result of combined sewer overflow from Hartford, Connecticut alone (CRWC 2006). The river has become much cleaner since the CWA was passed, but chromium, copper, nickel, lead, mercury, and zinc, chlordane, DDT, DDE, PCBs, and PAHs are found in quantities above the EPA recommended levels in sediments and fish tissue throughout the watershed (Jackson *et al.* 2005). Acid rain also affects rivers in the northeast, as it reduces the pH of rivers and causes metals to leach from bedrock at a faster rate (Usfws 2007).

Hydromodification Projects. The Connecticut River has 16 hydroelectric dams on the mainstem of the river and as many as 900 are estimated to have been built in the watershed. Fish ladders have been installed at Vernon, Turner Falls, and Holyoke Dams allowing fish passage to areas above Holyoke Dam in Massachusetts since 1981 (Usgs 2004). For some species, the ladders are not efficient, so fish passage continues to be compromised. For instance, overall passage efficiency at Turner Falls fish ladder is 17%, and has historically been inefficient at passing shad. Shortnose sturgeon are not able to migrate to spawning habitat above Holyoke Dam, which was recently re-licensed through 2039, so the only spawning shortnose sturgeon in the river are the fish that reside above the dam. The dams also affect the river's water quality, causing reduced DO and elevated water temperatures throughout the summer.

Mining. Dating back thousands of years, there is evidence of native people mining and extracting natural resources from the headwaters of the Connecticut River. There are many mines along the Connecticut River, which currently degrade the river's water quality, including the country's first chartered copper mine. Towns such as Plymouth, Vermont were famous for mining gold, iron, talc, soapstone, marble, asbestos, and granite (Ewald 2003). Other towns through New Hampshire and Vermont also mined gold, silver, soapstone, talc, granite, slate, and copper (Ewald 2003). In many locations, far downstream of the mines, accumulated heavy metals are in concentrations high enough to threaten aquatic life. In other cases, the mines are abandoned or failing and need to be cleaned. Such is the case with Elizabeth Mine, an old copper mine perched above the Connecticut River that leaches heavy metals into the river. As a result, Elizabeth Mine has been declared a superfund site. There is little to no mining in Long Island Sound and the concept is generally frowned upon in the region, although there has been and continues to be discussions about mining for sand and gravel.

Commercial and Recreational Fishing. There are not many commercial fisheries in the Connecticut River. Shad is the primary commercial fishery here, although shellfish, bluefish, striped bass, and flounder can be caught in the tidal estuary near the mouth. There are many recreationally angled fish, such as shad, striped bass, bluefish, northern pike, largemouth and smallmouth bass, perch, catfish, and other fish.

Long Island Sound fisheries provide an estimated 5.5 million dollars to the Connecticut economy. The primary fisheries target oysters, lobsters, scallops, blue crabs, flounder, striped bass, and bluefish. Recently, due to DO deficiencies, the western portion of Long Island Sound has seen major declines in fish and shellfish populations. Despite these recent declines, the sound houses the largest oyster fishery in the US, which provides 95% of the nation's oysters. At this same time, lobsters have been suffering from an unknown disease and their population has been declining. Simultaneously, menhaden have made a dramatic recovery over the past 10 years, which has resulted in much better fishing for larger predatory fish such as striped bass.

Directed harvest of shortnose sturgeon is prohibited by the ESA. However, shortnose sturgeon are likely taken incidentally in fisheries in the Connecticut River and Long Island Sound. Moser and Ross (1993) found that captures of shortnose sturgeon in commercial shad nets disrupted spawning migrations in the Cape Fear River, North Carolina, and Weber (1996b) reported that these incidental captures caused abandonment of spawning migrations in the Ogeechee River, Georgia. Entanglement of marine mammals in fishing gear is not uncommon and can lead to mortality or serious injury.

Hudson River Basin

Natural History. The Hudson River flows approximately 315 miles to the ocean, with a watershed of 13,365 mi^2. The river flows from the Adirondack Mountains, draining most of eastern New York State, to the ocean where the Hudson River canyon continues onto the continental shelf, marking where the original mouth of the Hudson was covered by rising sea levels after the last ice age. The Hudson River's bed is composed of metamorphosed plutonic rock in the Adirondack Mountains, then transitions to sedimentary rock, such as shale and limestone in the middle portion of the watershed, and the lower portion of the watershed is a mixture of sedimentary, metamorphic, and igneous rocks. The average precipitation is approximately 36 inches per year. At the mouth, the average discharge is 13.5 billion gallons each day, or 20,906 cubic feet per second (Jackson *et al.* 2005). The Hudson is a freshwater tidal estuary between Troy, NY at river mile 154 to Newburgh Bay at river mile 62, and then it is a tidal brackish estuary for the lower 62 miles to the Atlantic Ocean (Jackson *et al.* 2005). The river and estuary are home to over 200 fish species, with approximately 70 native freshwater fish species and 95 estuarine species having been recorded (Jackson *et al.* 2005).

Human Activities and their Impacts

Land Use. The Hudson River watershed usage is 25% agriculture, 65% forested, 8% urban, and 5% other (Jackson *et al.* 2005). Major towns in the Hudson River watershed are New York City, Albany, Poughkeepsie, and Hudson, New York and Jersey City, New Jersey. The human population in the watershed is approximately 350 people per square mile, but there are no people living in the headwaters and the population density in Manhattan is over 25,907 people per square mile (Jackson *et al.* 2005).

Throughout the 20th century, power plants, municipalities, pulp and paper mills, and corporations such as IBM, General Motors, and General Electric in particular, who the EPA estimates dumped between 209,000 and 1.3 million pounds of PCBs into the river, contributed large quantities of pollutants to the Hudson. The PCB levels in the Hudson River are amongst the highest nationwide. The upper basin is mostly unaffected by humans, with clear, soft water with low nutrients. The middle Hudson is more polluted, with 30 to 50% of the land in this region being used for agriculture and several cities such as Corinth, Glens Falls, Hudson Falls, and Fort Edward contributing industrial waste to the river. The tidal freshwater portion of the Hudson is nutrient rich with exceptionally low gradient. High tide in this stretch causes the river to flow backwards due to the low gradient and this prevents stratification. The brackish tidal estuary portion of the Hudson is nutrient rich with hard water. Two hundred miles of the Hudson River, from Hudson Falls to New York City, were designated as a superfund site due to the amount of pollution. There are still elevated amounts of cadmium, copper, nickel, chromium, lead, mercury, and zinc, DDT, PCBs, and PAHs are found in quantities above the EPA recommended levels in sediments and fish tissue throughout the watershed (Wall *et al.* 1998).

Hydromodification Projects. The mainstem Hudson River has 14 dams and there are dams near the mouths of many tributaries, but the lower 154 miles of tidally influenced river is undammed. Several flood control dams on tributaries such as the Indian and Sacandaga Rivers have drastically altered the flow of the mainstem Hudson River. The Hudson is an important river for anadromous fishes because it is unobstructed for the lower 154 miles, resulting in the healthiest population of ESA-listed endangered shortnose sturgeon in the United States. Prior to the Clean Water Act, the middle stretch of the Hudson and much of the lower reaches had low dissolved oxygen as a result of reduced flow behind the dams, high nutrients, and the collection of waste with high biological oxygen demand.

Mining. The Hudson River has been periodically important as a source of metals and mined resources. The Adirondack Mountains, in the headwaters, have mined silver, iron, titanium, coal, talc, vanadium, graphite, garnet, and zinc at various times over the past 300 years. McIntyre Mine is an example of a mine that has produced different minerals during different generations. Initially bought as an iron mine, McIntyre sat dormant for 75 years before titanium was discovered there, at which point National Lead purchased it and mined there until 1982 when NL Industries abandoned the mine.

Commercial and Recreational Fishing. The Hudson River commercial fishery historically caught fish, blue crabs, and oysters. Now, the only fish that is caught commercially in the Hudson is American shad. Historically, Atlantic sturgeon, striped bass, American eel, and white perch were productive commercial fisheries. The striped bass fishery closed in 1976 due to PCBs in the river and fish tissue. Atlantic sturgeon were fished until the mid 1990s. Blue crabs are still fished in the estuary all the way to Troy, NY with recent catches over 88,185 pounds per year. There is no commercial fishery for oysters but they used to be taken commercially in the brackish tidal section of the Hudson.

Delaware River Basin

Natural History. The Delaware River flows approximately 329 miles to the ocean, with a watershed of 12,757 mi^2. The river originates in the Catskill Mountains with over half of the river flowing through Pennsylvania and the rest of the watershed occupying parts of New Jersey, New York, and Delaware. The Delaware River's geology is sandstone with shale conglomerate in the upper watershed transitioning to sandstone, shale, and limestone in the middle watershed and igneous and metamorphic rock in the lower watershed. The average precipitation is approximately 43 inches per year. At the mouth, the average discharge is 9.6 billion gallons each day, or 14,903 cubic feet per second, and although it is only the 42nd largest river by discharge, Philadelphia is home to the largest freshwater port in the country (Jackson *et al.* 2005). The Delaware River estuary begins in Trenton, New Jersey and extends downstream for 144 miles (Jackson *et al.* 2005). The river and estuary are home to 105 species of fish, with approximately 8 nonnative fish (Jackson *et al.* 2005).

Human Activities and their Impacts

Land Use. The Delaware River watershed usage is 24% agriculture, 60% forested, 9% urban, and 7% surface water or other (Jackson *et al.* 2005). Major towns in the Delaware River watershed are Easton, Allentown, Reading, and Philadelphia, Pennsylvania; Trenton and Camden, New Jersey; and Wilmington, Delaware. The human population in the watershed is approximately 555 people per square mile (Jackson *et al.* 2005). The water quality was significantly degraded around Philadelphia by 1799. By the 1960s the average DO in the lower river was approximately 0.2ppm. A survey in the 1970s of organochlorine frequency in rivers ranked the Delaware at Trenton and the Schuylkill, the largest tributary to the Delaware, as the 8th and 1st worst, respectively in the nation (Jackson *et al.* 2005). While there aren't many point sources of pollution since the Clean Water Act was enacted, historically, power plants, municipalities, pulp and paper mills, and industries such as the Philadelphia Shipyard, Bethlehem Steel, New Jersey Zinc Company, contributed large quantities of pollutants to the Hudson. Approximately 95% of PCBs are introduced to the river through combined sewage overflows from treatment plants. Even 35 years after the Clean Water Act, there are still elevated amounts of copper, chromium, lead, mercury, and zinc, DDT, PCBs, and PAHs are found in quantities above the EPA recommended levels in sediments and fish tissue throughout the watershed (Wall *et al.* 1998). The heaviest concentrations of chemicals in the river occur in a 14 mile stretch between the Philadelphia naval yard and the Tacony-Palmyra Bridge.

Hydromodification Projects. The Delaware River has 16 dams in the headwaters but the middle and lower river is the longest undammed stretch of river east of the Mississippi. This stretch of free-flowing river is beneficial to anadromous and catadromous species, such as American shad, striped bass, and American eels.

Mining. The Delaware River watershed, particularly the eastern section was home to the majority of the nation's anthracite coal. As a result, many mining towns were established in the watershed to exploit the abundant resources. By 1914, over 181,000 people were employed as miners in the region. Apart from the coal mining, other minerals such as sulfur, talc, mica, aluminum, titanium, and magnesium were mined. Mines were also established for sand and gravel. Eventually minerals from the watershed were used to produce steel.

Commercial and Recreational Fishing. In the Delaware River, commercial fisheries exist for American shad, weakfish, striped bass, Atlantic croaker, Atlantic silversides, bay anchovy, black drum, hogchoker, northern kingfish and American eel. Commercial fishermen use gillnets and trawls as the primary means of capturing fish. Bycatch is a concern for the recovery of endangered shortnose sturgeon, where the highest mortality rates are recorded in gillnet fisheries. Recreational fishermen target weakfish, striped bass, croaker, drum, kingfish, and eel. No data exists on shortnose sturgeon poaching.

Chesapeake Bay Drainages

Natural History. Chesapeake Bay, the largest estuary in the United States, was formed by glacial activity more than 18,000 years ago. The Bay stretches some 200 miles from Havre de Grace, Maryland to Norfolk, Virginia, with more than 11,000 miles of shoreline. At its widest point, Chesapeake Bay is about 35 miles wide (near the Potomac River). Despite its massive size, the Bay is relatively shallow—average depth is only 21 feet—making it susceptible to significant fluctuations in temperature.

The Bay lies totally within the Atlantic Coastal Plain but the watershed includes parts of the Piedmont Province and the Appalachian Province. The tributaries provide a mixture of waters with a broad geochemical range to the Bay with its own mixture of minerals, nutrients and sediments depending on the geology of the place where the waters originate. In turn, the nature of the Bay itself depends on the characteristics and relative volumes of these contributing waters. While more than 50 tributaries deliver freshwater to Chesapeake Bay, major rivers include the Susquehanna, Potomac, and the James River, which we describe in greater detail below.

Susquehanna River. Ranked as the 18th largest river in the United States based on discharge, drainage area, or length, the Susquehanna River flows approximately 448 miles to the ocean, with a watershed of 27,580 mi^2 (Kammerer 1990, Jackson *et al.* 2005). The river flows north to south from New York, through Pennsylvania, and reaches the Chesapeake Bay in Havre de Grace, Maryland. The Susquehanna River's bed is rocky throughout, being described as a mile wide and a foot deep, with distinct pool/riffle formations even near the mouth. The average precipitation is approximately 39 inches per year. At the mouth, the average discharge is 26.3 billion gallons each day, or 40,718 cubic feet per second, and serves as the primary freshwater source of the Chesapeake Bay (Jackson *et al.* 2005). The Susquehanna isn't tidally influenced and doesn't have much estuary habitat (Jackson *et al.* 2005). The river is home to 103 fish species, but 27 of the fish are nonnative (Jackson *et al.* 2005).

Potomac River. The Potomac River is approximately 383 miles long and has a watershed of 14,670 mi^2. The river's headwaters begin in the Allegheny Mountains of West Virginia and the Potomac most famously flows through Washington, D.C., to the western side of the Chesapeake Bay. The substrate of the Potomac and its tributaries is mostly schist, phyllite, and metavolcanic rock. The average precipitation is approximately 39 inches. At the mouth, the average discharge is 7.3 billion gallons each day, or 11,301 cubic feet per second (Jackson *et al.* 2005). The Potomac River estuary begins two miles below the Washington, D.C. Maryland border, just below the Little Falls of the Potomac River. Ninety-five fish species live in the Potomac, but only 65 of those are native to the area (Jackson *et al.* 2005).

James River. The James River is approximately 340 miles long and drains a watershed of 10, 432 mi^2. The James River is one of the longest bodies of water in entirely in one state, beginning in the Allegheny Mountains of western Virginia and flowing across the state to the Chesapeake Bay. The upper James River's geology is primarily schist and siliclastic rock. The middle James River is primarily course grained conglomerates and sandstone. The lower section of the James is almost entirely sedimentary rock. The average precipitation is approximately 40 inches. At the mouth, the average discharge is 6.5 billion gallons each day, or 10,030 cubic feet per second (Blue 1998). The James River estuary begins at the fall-line in Richmond, Virginia. Ninety-five fish species live in the Potomac, but only 65 of those are native to the area (Jackson *et al.* 2005).

Human Activities and their Impacts

Land Use. The Susquehanna River watershed usage is 20% agriculture, 63% forested, 9% urban, and 7% pasture (Jackson *et al.* 2005). Major towns in the Susquehanna River watershed are Scranton, State College, and Harrisburg, Pennsylvania and Havre de Grace, Maryland. The human population in the watershed is approximately 145 people per square mile (Jackson *et al.* 2005). The water quality has not been well documented because the river wasn't used as a primary source of drinking water for any major cities. The three main events that had the greatest effect on the river were logging, dam building, and mining. While most of these activities took place in the 1800s, the river is still responding to the disruption they caused (Jackson *et al.* 2005). Sediment transport in the early 1900s was nine times higher than it was 200 years earlier, due to logging and agriculture. Sediment transport and its associated nutrients remain a major concern for the Chesapeake Bay. Coal is abundant through the watershed, amounting to nearly 30 billion tons of coal mined. Coal waste and acid mine drainage damaged much of the river and its tributaries. There was so much coal silt in the Susquehanna at one point that a fleet of over 200 vessels began harvesting the silt from the river's bed. From 1920 to 1950, over 3 million tons of coal were harvested from behind one dam. Later, between 1951 and 1973, over 10 million tons were harvested from behind another dam. Coal is no longer a primary industry in the watershed, but the impacts of the acid mine drainage are still prominent. Another major problem is untreated sewage and industrial waste that is dumped directly into the river. In Binghampton, New York, there are 10 sewer outfalls, 70 in Scranton, Pennsylvania, 65 in Harrisburg, Pennsylvania, and the number of outfalls totals over 400 in the watershed, generally with the number of outfalls being proportional to the size of the city. As a result, the Susquehanna contributes 44% of the nitrogen and 21% of the phosphorous to the Chesapeake Bay. This has led to large algal blooms in the bay and a resulting "dead zone" between Annapolis, Maryland and Newport News, Virginia. In 2005, the Susquehanna was named America's most endangered river by American Rivers, who produce an annual list. Even 35 years after the Clean Water Act, there are still elevated amounts of copper, sulfur, selenium, arsenic, cobalt, chromium, lead, mercury, zinc, and pesticides (Beyer and Day 2004).

The Potomac River watershed usage is 32% agriculture, 58% forested, 5% developed, 4% water, 1% wetland, and 1% barren (Jackson *et al.* 2005). Major towns in the Potomac River watershed are Washington, D.C.; Arlington and Alexandria, Virginia; and Hagerstown, Maryland. The human population in the watershed is approximately 358 people per square mile (Jackson *et al.*

2005). The water quality has significantly improved over the past 50 years. Even 35 years after the Clean Water Act, there are still elevated amounts of cadmium, chromium, copper, lead, dioxin, PCBs, and chlordane, which may have resulted in recent highly publicized reports of male fish producing eggs.

The James River watershed usage is 23% agriculture, 71% forested, and 6% urban (VDCR 2006). Major towns in the James River watershed are Charlottesville, Richmond, Petersburg, and Hampton Roads, Virginia. The human population in the watershed is approximately 2.5 million people, or approximately 240 people per square mile (VDCR 2006). The James River has 21 municipal dischargers permitted and 28 permitted industrial dischargers. There are also 18 EPA Superfund sites along the river, mostly found in the major cities along its corridor. In some cases, industries such as Allied Chemical were fined and forced to clean up large areas of extreme toxicity. Even 35 years after the Clean Water Act, there are still elevated amounts of zinc, copper, cadmium, nickel, chromium, lead, arsenic, dioxin, PCBs, and pesticides.

Hydromodification Projects. There are many dams along the Potomac River and its tributaries, but only three impoundments are larger than 1.5 square miles. One of the major tributaries, the Anacostia River, is having over 60 dams removed or altered to improve water quality and fish passage.

The Susquehanna River has over 100 dams along the mainstem and the first major dam is located just 10 miles upstream of the mouth. In recent years modern fishways have been installed in some of these dams and migratory fish appear to be responding positively. For instance, between 1928 and 1972, no shad passed Conowingo Dam, 10 miles upstream of the mouth of the Susquehanna River, but since fish began coming back, their abundance has increased from approximately 100 to more than 100,000.

The James River has several large dams along its length. Many dams have been removed or improved to allow fish passage, and in 1999, a ladder was built over Boscher Dam, which had prevented upstream fish runs since 1823. That ladder provided access to 137 additional miles of the James and 168 miles of its tributaries.

Mining. In the Chesapeake Bay watershed, coal mining has likely had the most significant impact on water quality. Mining in this watershed was so extensive that while many mines have been reclaimed and others are currently being reclaimed, at the current level of funding, it will take decades or more to completely reclaim all of the old mines in the watershed. Abandoned coal mines leach sulfuric acid as a result of natural reactions with the chemicals found in coal mines. Many of these abandoned coal mines must be treated with doses of limestone to balance the pH of the water draining from the mines. Much of the Appalachian Mountain chain that was mined for coal is now leaching sulfuric acid into tributaries of the Chesapeake Bay and requires some sort of treatment to improve the water quality of the region.

Commercial and Recreational Fishing. The Chesapeake Bay supports fisheries for American eel, croaker, blue crab, black sea bass, bluefish, oyster, red drum, spot, striped bass, summer flounder, weakfish, menhaden, and white perch (CFEPTAP 2004). Stocks of striped bass got so low in the mid 1980s that a moratorium started in 1985, but they recovered so well that well-regulated harvests are now permitted. Since the mid 1990s, levels of blue crab and menhaden have dropped to the lowest levels in history. Species such as catfish and white perch are year round residents and managed by individual states around the bay. Species like Spanish mackerel, king mackerel, red drum, and summer flounder have ranges that extend beyond the bay and are managed under multiple regional management plans. Some species such as American shad are allowed to be fished by some states (Virginia and Maryland) within the Chesapeake Bay, but not by other states (Delaware and Pennsylvania).

Ballast Water Bioregion CAR-VII (North Carolina to Florida)

This region covers all the drainages that ultimately drain to the Atlantic Ocean between the states of North Carolina and Florida. This region includes all of South Carolina and parts of Georgia, North Carolina, Florida, and Virginia. The region encompasses three ecoregions—the hot continental division, subtropical division, and savanna division (southern most tip of Florida’s panhandle). The hot continental division is characterized by it’s winter deciduous forest dominated by tall broadleaf trees, soils rich in humus and moderately leached (Inceptisols, Ultisols, and Alfisols), and rainfall totals that decrease with distance from the ocean (Bailey 1995).

Most of the Southeast Atlantic Coast Region is contained within the subtropical ecoregion and is characterized by a humid subtropical climate with particularly high humidity during summer months, and warm mild winters. Soils are strongly leached and rich in oxides of iron and aluminum (Bailey 1995). The subtropical ecoregion is forested, largely by second growth forests of longleaf, loblolly and slash pines, with inland areas dominated by deciduous trees. Rainfall is moderate to heavy with annual averages of about 40 inches in the north, decreasing slightly in the central portion of the region, and increasing to 64 inches in southern Florida. The savanna ecoregion has a tropical wet-dry climate, controlled by moist warm topical air masses and supports flora and fauna that is adapted to fluctuating water levels (Bailey 1995).

In the sections that follow we describe several basins and estuaries to characterize the general ecology and natural history of the area, and past and current human activities and their impacts on the area. The region contains more than 22 river systems that generally flow in a southeasterly direction to the Atlantic Coast. The diverse geology and climate ensures variability in biological productivity and hydrology. Major basins include the Albemarle-Pamlico Watershed and its tributaries, the Cape Fear River, Winyah Bay and the Santee-Cooper Systems, the Savannah, Ogeechee, and the St. Johns River, to name a few. The more northerly river, the Roanoke which is part of the Albemarle-Pamlico Watershed, is cooler and has a higher gradient and a streambed largely characterized by cobble, gravel and bedrock.

The southern rivers are characterized by larger portions of low gradient reaches, and streambeds that are composed of greater amounts of sand and fine sediments—are often high in suspended solids, and have neutral to slightly acidic waters with high concentrations of dissolved organic carbon. Rivers emanating entirely within the Coastal Plain are acidic, low alkalinity, blackwater systems with dissolved organic carbon concentrations often up to 50 mg/L (Smock and Benke 2005). We described several river basins in detail to provide additional context for evaluating the influence of the environmental baseline on listed species under NMFS' jurisdiction and the health of the environment.

Albemarle-Pamlico Sound Complex

Natural History. The Albemarle-Pamlico Sound Estuarine Complex, the largest lagoonal estuarine system in the United States, includes seven sounds including Currituck Sound, Albemarle Sound, Pamlico Sound and others (EPA 2006b). The Estuarine Complex is separated from the Atlantic Ocean by the Outer Banks, a long barrier peninsula, and is characterized by shallow waters, wind-driven tides that result in variable patterns of water circulation and salinity. Estuarine habitats include salt marshes, hardwood swamp forests, and bald cypress swamps.

The Albemarle-Pamlico watershed encompasses four physiographic regions—the Valley and Ridge, Blue Ridge, Piedmont and Costal Plain Provinces. The geology of the basin strongly influences the water quality and quantity within the basin. The headwaters of the basin tributaries are generally steep and surface water flowing downstream has less opportunity to pick up dissolved minerals. However, as the surface water flows reaches the Piedmont and Coastal Plain, water velocity slows due to the low gradient and streams generally pick up two to three times the mineral content of surface waters in the mountains (Spruill and Survey 1998). At the same time, much of the upper watershed is composed of fractured rock overlain by unconsolidated and partially consolidated sands. As a result, of the basin's geology, as a general matter more than half of the water flowing in streams discharging to the Albemarle-Pamilico Estuarine Complex comes from ground water.

Primary freshwater inputs to the Estuary Complex include the Pasquotank, Chowan and Roanoke Rivers that flow into Albemarle Sound, and the Tar-Pamlico and Neuse Rivers that flow into Pamlico Sound. The Roanoke River is approximately 410 miles long and drains a watershed of 9,580 mi^2. The Roanoke River begins in the mountains of western Virginia and flows across the North Carolina border before entering the Albemarle Sound. The upper Roanoke River's geology is primarily a high gradient boulder-rubble bedrock system. The middle Roanoke River is primarily course sand and gravel. The lower section of the Roanoke is almost entirely organic-rich mud. The average precipitation is approximately 43 inches. At the mouth, the average discharge is 5.3 billion gallons each day, or 8,193 cubic feet per second (Smock and Benke 2005). The Roanoke River is home to 119 fish species, and only seven of those are not native to the area (Smock and Benke 2005). The Roanoke is also home to nine endangered fish species, two amphibians, and seven mussels, including several important anadromous fish species.

The Neuse River is 248 miles long and has a watershed of 6,235 mi^2 (Smock and Benke 2005). The Neuse River watershed is also located entirely within the state of North Carolina, flowing through the same habitat as the Cape Fear River, but ultimately entering Pamlico Sound. The river originates in weathered crystalline rocks of the piedmont and crosses sandstone, shale, and limestone before entering Pamlico Sound (Turekian *et al.* 1967). The average precipitation is approximately 48 inches. At the mouth, the average discharge is 3.4 billion gallons each day, or 5,297 cubic feet per second (USGS 2005).

Human Activities and their Impacts

Land Use. Land use in the Roanoke River is dominated by forest (68%) and the basin contains some of the largest intact, least disturbed bottomland forest floodplains along the eastern coast. Only 3% of the basin qualifies as urban land uses, and 25% is used for agriculture (Smock and Benke 2005). The only major town in the Roanoke watershed is Roanoke, Virginia. The population in the watershed is approximately 80 people per square mile (Smock and Benke 2005). In contrast, the Neuse River watershed is described as 35% agriculture, 34% forested, 20% wetlands, and 5% urban, and 6% other, with a basin wide density of approximately 186 people per square mile (Smock and Benke 2005). While the population increased in the Albemarle-Pamlico Complex more than 70% during the last 40 years, the rate of growth is relatively low for many coastal counties in the Southeast (EPA 2006b). Much of the estuarine complex is protected by large amounts of state and federally protected lands, which may reduce development pressures.

Throughout the 20th century, mining, agriculture, paper and pulp mills, and municipalities contributed large quantities of pollutants to the Roanoke River and the Albemarle-Pamlico Estuarine Complex. Even so, today the Albemarle-Pamlico Estuarine Complex is rated in good to fair condition in the National Estuary Program Coastal Condition Report despite that over the past 40-year period data indicate some noticeable changes in the estuary, including increased dissolved oxygen levels, increased pH, decreased levels of suspended solids, and increased chlorophyll *a* levels (EPA 2006b).

Coal is mined from the mountainous headwaters of the Roanoke River in southwestern Virginia. Mining through the piedmont and coastal areas of North Carolina was conducted for limestone, lead, zinc, titanium, apatite, phosphate, crushed stone, sand, and fossils. Many active mines in these watersheds are still in operation today. These mines are blamed for increased erosion, reduced pH, and leached heavy metals.

Agricultural activities are major source of nutrients to the estuary and a contributor to the harmful algal blooms (HABs) in summer, although according to McMahon and Woodside 1997 (EPA 2006a) nearly one-third of the total nitrogen inputs and one-fourth of the total phosphorus input to the estuary are from atmospheric sources. Primary agricultural activities within the watershed include corn, soybean, cotton, peanut, tobacco, grain, potato, and the production of chicken, hog, turkey, and cattle.

In general, the Roanoke River is much cleaner since the passage of the CWA, although mercury, arsenic, cadmium, chromium, copper, lead, nickel, zinc, and PCBs are still considered high (NCDENR 1999). Fish tissues sampled within the estuary also showed elevated concentrations of total PAHs and total PCBs—10% of the sampled stations exceeded risk-based EPA Advisory Guidance values (EPA 2006b). Water quality studies in the mid-1990s showed the Neuse Basin contained the highest nitrogen and phosphorus yields, while the Chowan Basin had the lowest yields (Spruill and Survey 1998).

The Neuse River entered the national spotlight during the early 1990s due to massive and frequent fish kills within the basin. Over one billion American shad have died in the Neuse River since 1991. The problem is persistent but the cause of the kills differs among events; in 2004 more than 700,000 estuarine fish died and more than 5,000 fresh fish died within the basin.

Freshwater species most commonly identified during investigations included sunfishes, shad, and carp, while estuarine species most commonly reported included menhaden, perch, and croaker. Atlantic menhaden have historically been involved in a majority of estuarine kill events and have exhibited stress and disease in conjunction with fish kills. Fish kill events may often have different causative agents, and in many cases the precise cause is not clear, but high levels of nutrients, HABs, toxic spills, outbreaks of a marine organism, *Pfiesteria pescicida*, low DO concentrations and sudden wind changes that mix hypoxic waters, are some of contributing factors or causes to the basins persistent fish kills (NCDEQ 2004).

Both the Roanoke River and the Neuse Rivers are fragmented by dams. The reservoirs are used for flood control and recreation, but the amount of agricultural and urban runoff that collects behind the dams has caused sanitation problems in the recent past. Three dams were removed recently in an effort to improve environmental conditions and fish passage. Widespread stream modification and bank erosion were rated high within the greater watershed relative to other sites in the Nation (Spruill and Survey 1998).

Commercial and Recreational Fishing. The Albemarle and Pamlico Sounds and associated rivers support a dockside commercial fishery valued at over $54 million annually. The commercial harvest includes blue crabs, southern flounder, striped bass, striped mullet, white perch, croaker, and spot, among others. Roughly 100 species are fished commercially or recreationally in the region. The Neuse River supports many of the same species as the Roanoke River.

Commercial and recreational fisheries exist for oyster, crab, clam, American shad, American eel, shrimp, and many other species. Shellfish can be collected by dredging, which has adverse effects to benthic organisms, including shortnose sturgeon that use estuarine areas for feeding. Commercial fisheries along the South Carolina coast use channel nets, fyke nets, gillnets, seines, and trawls. All of those methods must use some sort of turtle excluder device, but could still accidentally capture a shortnose sturgeon.

Major Southeast Coastal Plains Basins

Natural History. More than five major river basins flow through the Coastal Plains of the Southeast and directly enter the Atlantic Ocean including the Cape Fear, Great Pee-Dee, Altamaha, and the St. Johns Rivers (see Table 30 for a description of several basins within this region). Rainfall is abundant in the region and temperatures are generally warm throughout the year. Northern rivers originate in the Blue Ridge Mountains or the Piedmont Plateau, but all the rivers described in this section have sizeable reaches of slack water as they flow through the flat Coastal Plain. Two rivers, The Satilla River in Georgia and the St. Johns River in Florida, are located entirely within the Coastal Plain. The highest elevation of the St. Johns River is 26 feet above sea level, so the change in elevation is essentially one inch every mile, making it one of the most gradually flowing rivers in the country.

Smock *et al.* (2005) describe the mountains and plateau as areas of heavily dissected and primarily highly metamorphosed rock of Paleozoic age, with occasional areas of igneous and sedimentary rock. Underlying rock is varied with bands of limestone, dolomite, shale, sandstone, cherts, and marble, with a number of springs and caves scattered throughout the area.

Where the Piedmont Plateau dips the sedimentary deposits of the coastal plain is termed the fall line. Here, steep changes in elevation result in rapids or falls before the rivers level off in their Coastal Plain reaches. In the Coastal Plain reaches of the areas rivers soils are acidic with a low cation exchange capacity and a sandy or loamy surface horizon, and a loamy or clay subsurface. The acidic characteristics, slow flowing water with poor flushing and high organic and mineral inputs gives these waters their characteristic “blackwater” (or “brownwater” for those that originate in the Piemont Plateau) appearance. The Satilla River is a blackwater river that has a naturally low pH (between 4 and 6) and white sandbars--due to the low pH it also has naturally lower productivity than other rivers that originate within the mountains or the Plateau.

Table 30. General Information about Rivers of the Southeast United States (NCDENR 1999, Smock and Benke 2005).

Watershed	Length (mi.)	Basin Size (mi^2)	Physiographic Provinces*	Mean Annual Precipitation (in.)	Mean Discharge (cfs).	No. Fish Species	No. Endangered Species
Cape Fear River	320	9,324	PP, CP	47	7,663	95	8 fish, 1 mammal, 15 mussels
Great Pee Dee River	430	10,641	BR, PP, CP	44	13,102	>100	6 fish, 1 reptile
Santee-Cooper River	440	15,251	BR, PP, CP	50	15,327	>100	5 fish, 2 reptiles
Savannah River	300	10,585	BR, PP, CP	45	11,265	>100	7 fish, 4 amphibians, 2 reptiles, 8 mussels, 3 crayfish
Ogeechee River	250	5,212	PP, CP	44	4,061	>80	6 fish, 2 amphibians, 2 reptiles, 1 mussel
Altamaha River	140 (>400)	14,517	PP, CP	51	13,879	93	1 mammal, 12 fish, 2 amphibians, 2 reptiles, 7 mussels, 1 crayfish
Satilla River	200	3,530	CP	50	2,295	52	2 fish, 1 amphibian, 2 reptiles, 1 mussel
St. Johns River	311	8,702	CP	52	7,840	>150	1 mammal, 4 fish, 2 reptiles, 2 birds

* Physiographic Provinces: BR = Blue Ridge, PP = Piedmont Plateau, CP = Coastal Plain

Human Activities and their Impacts

Land Use. Across this region, land use is dominated by agriculture and industry, and to a lesser extent timber and paper production, although more than half of most basins remain forested. Basin population density is highly variable throughout the region with the greatest density in the St. Johns River watershed with about 200 people per square mile of catchment, most of whom are located near Jacksonville, Florida. In contrast, there are only 29 people per square mile in the Saltilla River watershed in Georgia (Smock and Benke 2005). See Table 31 for a summary of land uses and population densities in several area basins across the region (data from (Smock and Benke 2005).

The largest population centers in the region include Miami and Jacksonville, Florida, and Savannah, Georgia. Major towns include Greensboro, Chapel Hill, Fayetteville, South Carolina, and Wilmington, North Carolina in the Cape Fear River watershed; Winston-Salem, North Carolina and Georgetown, Florence, and Sumter, South Carolina in the Great Pee-Dee River Watershed; Charlotte, Hickory, and Gastonia, North Carolina and Greenville and Columbia, South Carolina in the Santee-Cooper River watershed; Savannah and Augusta, Georgia, in the Savannah River watershed; Louisville, Statesboro, and Savannah, Georgia, in the Ogeechee

River watershed; Athens, and Atlanta, Georgia, in the Altamaha River watershed; and Jacksonville, Florida in the St. Johns River watershed.

Several of the rivers in the region have elevated levels of metals including mercury, fecal coliform, bacteria, ammonia, turbidity, and low DO. These impairments are caused by municipal sewage overflows, mining, and non-point source pollution, waterfowl, urban runoff, marinas, agriculture, and industries including textile manufacturing, power plant operations, paper mills and chemical plants (Harned and Meyer 1983, Berndt *et al.* 1998, NCDWQ 1998, Smock and Benke 2005).

Several watersheds exhibit high nitrogen loads including the Cape Fear River, Winyah Bay, Charleston Harbor, St. Helena Sound, Savannah River, Ossabaw Sound, Altamaha River, and St. Mary's River and Cumberland Sound (Bricker *et al.* 2007). Nitrate concentrations (as nitrogen) tend to be higher in stream draining basins with agricultural and mixed land uses (Berndt *et al.* 1998). Based on studies in Georgia, however, nitrate loads did not vary with growing season of crops (periods of heaviest fertilizer application), but were influenced by high streamflow, which could be related to downstream transport by subsurface flows (Berndt *et al.* 1998).

Table 31. Land Uses and Population Density in Several Southeast Atlantic Basins (Smock and Benke 2005)

Watershed	Land Use Categories (Percent)				Density (people/mi.^{2})
	Agriculture	Forested	Urban	Other	
Cape Fear River	24	56	9	11	80
The Great Pee-Dee	28	58	8	6	127
Santee-Cooper River	26	64	6	4	168
Savannah River	22	65	4	9	91
Ogeechee River	18	54	1	17 (wetlands)	78
Altamaha River	--	64	3	7	73
Satilla River	26	72	1	1	29
St. Johns River	25	45	6	24 (wetlands & water)	202

Sediment is the most serious pollutant in the Yadkin (Pee-Dee) River and has historically been blamed on agricultural runoff. In the mid 1990s, farmers in the region began using soil conservation techniques that have reduced sediment inputs by 77%. Unfortunately, the reduction in sediment inputs from farms did not translate to a reduction in sediment in the river, as during this period there was a 25% reduction in agricultural land and a 38% increase in urban development.

Mining. Mining occurs throughout the region. South Carolina is ranked 25th in the states in terms of mineral value and 13th among the eastern 26 states, and produces 1% of the total nonfuel mineral production value in the United States. There are currently 13 minerals being extracted from 485 active mines in South Carolina alone. Portland and masonry cement and crushed stone were the State's leading nonfuel minerals in 2004 (NMA 2007). In contrast, Georgia accounts for 4%, Florida accounts for 5%, and North Carolina accounts for 1.76% of the total nonfuel mineral production value in the United States. North Carolina's leading nonfuel minerals in 2004 were crushed stone, phosphate rock, and construction sand and gravel. Georgia produces 24% of the clay in the nation; other leading nonfuel minerals include crushed stone and Portland cement. Florida is the top phosphate rock mining state in the United States and produces about six times more than any other state in the nation. Peat and zirconium concentrates are also produced in Florida.

The first gold mine discovered and operated in the United States is outside Charlotte, North Carolina in the Pee Dee watershed. Mines through Georgia are also major producers of barite and crude mica, iron oxide, and feldspar. There is a proposed titanium mine near the mouth of the Satilla River. Unfortunately, mines release some toxic materials and negatively impact fish, as fish living around dredge tailings have elevated levels of mercury and selenium.

Hydromodification Projects. Several of the rivers within the area have been modified by dams and impoundments. In contrast to rivers along the Pacific Coast, we found considerable less information on other types of hydromodification projects in this area, such as levees and channelization projects. There are three locks and dams along the mainstem Cape Fear River and a large impoundment on the Haw River. The lower river and its tributaries are relatively undisturbed. The lower reach is naturally a blackwater river with naturally low dissolved oxygen, which is compounded by the reduced flow and stratification caused by upstream reservoirs and dams. The Yadkin (Pee Dee) River is heavily utilized for hydroelectric power. There are many dams on Santee-Cooper River System. The Santee River Dam forms Lake Marion and diverts the Santee River to the Cooper River, where another dam, St. Stephen Dam, regulates the outflow of the Santee River. Lake Moultrie is formed by both St. Stephen Dam and Pinopolis Dam, which regulates the flow of the Cooper River to the ocean. Below the fall line, the Savannah River is free-flowing with a meandering course, but above the fall line, there are three large dams that turn the piedmont section of the river into a 100-mile long stretch of reservoir. Although the Altamaha River is undammed, hydropower dams are located in its tributaries the Oconee and Ocmulgee Rivers above the fall lines. There are no dams, however, along the entire mainstem Satilla River. There are no major dams on the mainstem St. Johns River either, but one of the largest tributaries has a dam on it. The St. Johns River's flow is altered, however, by water diversions for drinking water and agriculture.

Commercial Fishing. The region is home to many commercial fisheries targeting species like shrimp, blue crab, clams, American and hickory shad, oysters, whelks, scallops, channel catfish, flathead catfish, snapper, and grouper. Shortnose sturgeon can be caught in gillnets, but gillnets and purse seines account for less than 2% of the annual bycatch. Shrimpers are responsible for 50% of all bycatch in Georgia waters and often interact with sea turtles. There are approximately 1.15 million recreational anglers in the state.

Ballast Water Bioregion CAR-I (Florida to Texas)

This region encompasses states of Alabama, Arkansas, Illinois, Iowa, Kansas, Kentucky, Louisiana, Mississippi, Missouri, Oklahoma, South Dakota, Tennessee, the western portion of Florida including the Florida Keys, and parts of, Georgia, Texas, Minnesota, Montana, North Dakota, Nebraska, Colorado, Indiana, Ohio, New Mexico, North Carolina, Pennsylvania, Virginia, West Virginia, Wisconsin, Wyoming, Mexico, and two Canadian provinces. Almost

2/3 of the continental United States drains to the Gulf of Mexico through the Mississippi River Basin.

While the Mississippi River is the most notable basin that drains to the Gulf of Mexico in terms of overall size (and the largest river in the United States) more than ten major river basins flow through to the Gulf including the Atchafalaya, Mobile, Red, Brazos, Colorado, and Rio Grande Rivers several (see Table 34 for a description of several basins within this region). In the following sections, we describe several basins and estuaries that enter the Gulf of Mexico to characterize the general ecology and natural history of the area, and past and current human activities and their impacts on the area.

Natural History. Due to the enormity of the drainages in this region, several ecoregions are encompassed in this region including the subtropical, the tropical/subtropical steppe, hot continental and mountain segments, temperate steppe, and the prairie ecoregions (Bailey 1995). Most of the region is within the subtropical ecoregion (division) and is characterized by a humid subtropical climate with particularly high humidity during summer months and warm mild winters. Soils are strongly leached and rich in oxides of iron and aluminum (Bailey 1995). The region is forested, largely by second growth forests of longleaf, loblolly and slash pines with inland areas dominated by deciduous trees. Rainfall is moderate to heavy with annual averages of about 40 inches in the north, decreasing slightly in the central portion of the region, and increasing to 64 inches in southern Florida.

The geology of the eastern Gulf Coast is primarily sedimentary rocks of both siliclastic (sand, silt, clay) and carbonate (limestone and dolomite) types. Karst is a major mineral in Florida. The piedmont region of Georgia is composed of metamorphosed sedimentary rock and overlaid by decomposed rock called saprolite. Saprolite is rich in aluminum, silicon, and iron oxide. The metamorphosed sedimentary rock is also rich in minerals that intruded during earthquakes millions of years before.

Soils in the eastern Gulf are rich in oxides of iron and aluminum, moister and strongly leached (Bailey 1995), whereas soils in the western Gulf Coast highly varied, and reflect climate and geological differences. Arid parts of the region exhibit calcarious and/or gypsum-rich soils, and tend to have a neutral pH, whereas prairie soils are commonly slightly acidic sandy to clay loams. There is a strong decline in total rainfall moving east to west, which strongly affects vegetation patterns, river discharge (see Table 32 – rivers are listed in their general east to west pattern).

Table 32. Select Rivers in the Gulf Coast Region (Kammerer 1990, Brown *et al.* 2005, Dahm *et al.* 2005, Ward and Ward 2005)

Watershed	Length (mi. [approx.)	Basin Size (mi^2)	Physiographic Provinces*	Mean Annual Precipitation (inches)	Mean Discharge (cfs).	No. Fish Species	No. Endangered Species
Suwanee River	245	9,640	CP	53	10,804	81	1 fish
Apalachicola River System	106 (>530)	19,571	BR, PP, CP	50	26,804	104	1 fish, 1 reptile
Choctawhatchee River	170	4,646	CP	57	7,487	80	1 fish
Escambia-Conecuh River	231	4,233	CP	65	6,922	102	0

Watershed	Length (mi. [approx.)	Basin Size (mi^2)	Physiographic Provinces*	Mean Annual Precipitation (inches)	Mean Discharge (cfs).	No. Fish Species	No. Endangered Species
Mobile River	774	43,000	CP, VR, AP, PP, BR	50	67,592	236	12 fish, 3 reptiles, 19 mussels, 7 snails
Pascagoula River	140 (>400)	9,498	CP	61	15,256	119	1 fish, 2 reptiles
Pearl River	409	8,494	CP	56	13,172	119	1 fish, 2 reptiles, 1 mussel
Mississippi River	2,320	1,151,000		39	450,000	375	
Sabine River	555	9,756	CP	50	8,405	>104	>4 fish, 2 crayfish
Neches River	416	10,011	CP	54	6,321	96	≥4 fish, 1 crayfish
Trinity River	550	17,969	CL, GP, CP	45	7,840	99	3 fish, 1 crayfish, 3 mussels
Brazos River	1,280	44,620	CL, GP, CP	32	8,793	93	>4 fish, 4 mussels
Colorado River	862	39,900	CL, GP, CP	32	2,649	98	>4 fish, 2 salamanders, 1 snake, 5 mussels
San Antonio/Guadalupe Rivers	408	10,128	GP, CP	32	2,790	88	≥ 7 fish, several amphibians, 3 spring/cave pool-associated aquatic insects, 1 plant
Nueces River	315	16,800	GP, CP	24	706	≥66	≥3 fish
Rio Grande	1,759	335,908	SR, CO, B/R, GP, CP, SC, SO	8	1,307	>160	≥16 fish, several mollusks, 6 birds

Physiographic Provinces: BR = Blue Ridge, PP = Piedmont Plateau, CP = Coastal Plain, VR=Valley Ridge, AP=Appalachian Plateau, SR=Southern Rock Mtns., CO=Colorado Plateau, B/R=Basin & Range, GP=Great Plains, SC=Sierra Madre Occidental, SO=Sierra Madre Oriental, CL=Central Lowlands

Human Activities and their Impacts

Land Use. Land use is dominated by forest in the basins east of the Mississippi, whereas grass/shrub and rangeland uses dominate in basins west of the Mississippi. The Mississippi also appears to be a divide between the less developed eastern basins, and the increasingly urbanized western basins. According to data presented in Table 33, the most developed watersheds are the Trinity River, the San Antonio and Guadalupe Rivers, the Brazos River, the Colorado River, and the Mississippi River. Most of the population within the San Antonio River watershed is concentrated within the greater San Antonio area. Based on data from 2000, the population density of San Antonio is an estimated 1,122 people/mi^2, and in other areas of the basin density is as little as 16 people/mi^2 (Dahm *et al.* 2005). The Trinity River Basin encompasses several urban areas including one of the most highly populated areas in the region--the City of Dallas. In stark contrast, overall there are only 29 people per square mile in the Neches River watershed (Dahm *et al.* 2005).

Major threats to the southwestern basins also include wastewater effluent, water extraction, non-point source pollution, nonnative species, existing impoundments, and proposals for dams (Dahm *et al.* 2005), and new reservoirs are proposed for some basins (Lane-Miller and DeVries 2007). Municipal waste water discharge poses a serious problem in several rivers, including the Suwannee River basin, and the Chattahoochee and Flint Rivers. According to Dahm *et al.* (2005) the Rio Grande is one of the most impacted rivers due to water quality and quantity concerns. The basin suffers from elevated levels of salinity, nutrients, bacteria, metals, pesticides, herbicides, organic solvents, and the basin is heavily hydromodified by dams and water diversions for irrigation. About 100 miles downstream of Atlanta the Chatahoochee is very polluted, with excessive amounts of nutrients, pesticide, fecal coliform bacteria, PAHs, and oils. The lower Mississippi River is degraded by excess fecal coliform bacteria, PCBs, chlordane, turbidity, siltation, nutrients, reduced DO, pesticides, and eutrophication. Most of the riparian habitat has been lost to agriculture and urban development (Brown *et al.* 2005).

In many basins agricultural practices associated with row crops (corn, soybeans, hay and cotton) confined animal feeding operations (poultry and lifestock—hog, cattle, sheep, goats), and dairy production are significant source of nutrients, fecal coliform, and pesticides. Other basins are severely impacted by altered sediment regimes. The Choctawhatchee River watershed has highly erodable soils, heavy rains, and intermittent droughts that leads to excessive sediment loading. Erosion causes sediment and nutrient issues, while droughts cause low flow and low dissolved oxygen. In contrast, downcutting of reaches of the Brazos River are a problem resulting from numerous dams interrupting sediment transport within the basin.

Several rivers including the Pascagoula River and its tributaries, and the Sabine River are also impaired by sediment, pathogens, low DO, fecal coliform, nutrients, mercury, PCB, dioxin, ammonia, pesticides like atrazine, and BOD. Occasional fish kills occur within the Colorado River as a result of storm runoff and low DO. The upper Colorado River has salinity problems and many reservoirs have problems with toxic golden algae (Dahm *et al.* 2005). The upper Brazos River basin has naturally high salinity, the middle basin has elevated nutrients from nearby dairy farms, several reservoirs have toxic golden algae, and the lower basin has elevated atrazine, bacteria, phosphorous, and low DO (Dahm *et al.* 2005). Major polluters in the Mobile River include pulp and paper mills, textiles, chemical plants, hydroelectric, iron and steel manufacturing, and coal plants.

Pollution of this nature can reduce productivity and health of the fish populations within the basin, and at times can lead to fish kills. Since 1998, there have been at least 16 fish kills, at least one of which was the result of elevated ammonia levels, two were contributed to pesticides, 10 were from low DO, and 3 were from unknown causes (MSDEQ 2000). Large fish kills are the most severe and usually the most easily observed response of aquatic ecosystems to pollution, but often the degradation is more elusive occurring at sublethal levels.

Table 33. Land Uses and Population Density in Several Gulf of Mexico Basins (Brown *et al.* 2005, Dahm *et al.* 2005, Ward and Ward 2005).

Watershed	Land Use Categories (Percent)				Density (people/mi.^{2})
	Agriculture	Forested	Urban	Other	
Suwannee River	30	38	1	9	57
Apalachicola River System	25	55	2	18 (10% wetland)	133
Choctawhatchee River	25	57	1	17 (9% wetland)	46
Escambia-Conecuh River	15	72	<1	12 (7% wetland)	86
Mobile River	18	68	2	12 (7% wetlands)	114
Pascagoula River	17	66	1	16 (11% wetland)	75
Pearl River	24	58	2	15 (12% wetland)	109
Mississippi River	57	28	14	---	26
Sabine River	10	67	8	15 grassland	47

Watershed	Land Use Categories (Percent)				Density (people/mi.^{2})
	Agriculture	Forested	Urban	Other	
Neches River	15	65	5	15 grassland	29
Trinity River	15	35	30	20 grassland	254
Brazos River	24	3	16	15 grassland	52
Colorado River	30	--	15	55 range	91
San Antonio and Guadalupe Rivers	15	--	25	60 range	220
Nueces River	15	--	5	55 shrubland	42
Rio Grande River	5	14	7	74 shrub & grass	42

Mining. Mining occurs throughout the region. Mining along the eastern Gulf of Mexico coast is primarily for clay, sand, limestone, phosphate, and peat. There are also some sulfide mines upstream on the Apalachicola River and gravel mines in the Escambia River.

Hydromodification Projects. Several of the rivers within the area have been modified by dams, impoundments for navigation, levees, and drainage systems. Some rivers on both the eastern and western portion of the Gulf (including the Mississippi River) have been heavily hydromodified—fragmented by hydroelectric power plants and navigational dams, channels have been deepened, straightened, and contained within levees. For instance, there are 13 dams on the mainstem Chattahoochee and three on the Flint River, but there are no major dams on the Apalachicola River. There are 36 major dams in the Mobile River watershed, and the Trinity River watershed is also highly fragmented with 21 major dams throughout the watershed.

There are more than a 132 dams on the Brazos River—as a result of the dams there has been a reduction in sediment transport to reaches below the dams, consequently the river channel has deepened (downcut) resulting in the isolation of the mainstem from several of the oxbow lakes and off channel habitat once available to the native fishes and other animals. According to Dahm *et al.* (2005), although development is not prevalent in the lower river due to the frequency of flooding, the river is threatened by existing and proposed diversions to the neighboring cities of Houston and Fort Worth. Additionally, dredging activities have been documented to capture or kill 168 sea turtles from 1995 to 2009 in the Gulf of Mexico, including 97 loggerheads, 35 Kemp’s ridleys, 32 greens, and three unidentified sea turtles (USACOE 2010).

Commercial and Recreational Fishing. There is an extensive commercial fishery in the Gulf of Mexico. Fishermen fish with gillnets, trawls, paired trawls, and cast nets. Recreational fishermen are allowed to use hand lines, rod and reels, spears, and cast nets. This gear poses a risk to gulf sturgeon as a potential bycatch species. Gulf of Mexico fishing regulations require special gear to release turtles and smalltooth sawfish.

Ballast Water Bioregions NEP-VI (US/Mexico Border to Los Angeles) and NEP-V (California north of Los Angeles)

The basins described in this section empty into the coastal waters of California and are encompassed by the state of California and parts of Oregon. Select watersheds described herein characterize the general ecology and natural history of the area, and the past, present and future human activities and their impacts on the area. Essentially, this region encompasses all Pacific Coast Rivers south of Cape Blanco, California through southern California. The Cape Blanco area marks a major biogeographic boundary and has been identified by NMFS as a DPS/ESU boundary for Chinook and coho salmon, and steelhead on the basis of strong genetic, life history, ecological and habitat differences north and south of this landmark. Major rivers contained in this grouping of watersheds are the Sacramento, San Joaquin, Salinas, Klamath, Russian, Santa Ana and Santa Margarita Rivers see Table 34).

Natural History. The physiographic regions covered by the basins discussed herein, include: (a) the Cascade-Sierra Nevada Mountains province, which extends beyond this region as we have defined it and continue north into British Columbia, (b) the Pacific Border province, and (c) the Lower California province (Carter and Resh 2005). The broader ecoregions division, as defined by Bailey (1995) is the Mediterranean Division. Three major vegetation types are encompassed by this region: the temperate coniferous forest, the Mediterranean shrub and savannah, and the temperate grasslands/savannah/shrub. The area, once dominated by native grasses, is naturally prone to fires set by lightening during the dry season (Bailey 1995).

This region is the most geologically young and tectonically active region in North America. The Coast Range Mountains are folded and faulted formations, with a variety of soil types and nutrients that influence the hydrology and biology of the individual basins (Carter and Resh 2005). The region also covers the Klamath Mountains and the Sierra Nevada.

The climate is defined by hot dry summers and wet, mild winters, with precipitation generally decreasing in southern latitudes although precipitation is strongly influences by topograph and generally increases with elevation. Annual precipitation varies from less than 10 inches to more than 50 inches in the region. In the Sierra Nevada about 50% of the precipitation occurs as snow (Carter and Resh 2005), as a result snowmelt strongly influences hydrological patterns in the area. Severe seasonal patterns of flooding and drought, and high interannual variation in total precipitation makes the general hydrological pattern highly predictable within a basin, but the constancy is low across years (Carter and Resh 2005). According to Carter and Resh (2005) this likely increases the variability in the annual composition of the fish assemblies in the region (Table 34).

Table 34. Select Rivers in the Southwest Coast Region (Carter and Resh 2005)

Watershed	Length (mi. [approx.)	Basin Size (mi^2)	Physiographic Provinces*	Mean Annual Precipitation (inches)	Mean Discharge (cfs).	No. Fish Species (native)	No. Endangered Species
Rogue River	211	5,154	CS, PB	38	10,065	23 (14)	11
Klamath River	287	15,679	PB, B/R, CS	33	17,693	48 (30)	41
Eel River	200	3651	PB	52	7416	25 (15)	12
Russian River	110	1439	PB	41	2331	41 (20)	43
Sacramento River	400	27,850	PB, CS, B/R	35	23,202	69 (29)	>50 T & E spp.
San Joaquin River	348	83,409	PB, CS	49	4,662	63	>50 T & E spp.
Salinas River	179	4241	PB	14	448	36 (16)	42 T & E spp.
Santa Ana River	110	2438	PB	13	60	45 (9)	54

Watershed	Length (mi. [approx.)	Basin Size (mi^2)	Physiographic Provinces*	Mean Annual Precipitation (inches)	Mean Discharge (cfs).	No. Fish Species (native)	No. Endangered Species
Santa Margarita River	27	1896	LC, PB	49.5	42	17 (6)	52

* Physiographic Provinces: PB = Pacific Border, CS = Cascades-Sierra Nevada mountains, B/R=Basin & Range

The San Joaquin River, drains the largest basin in the region, originates within the Sierra Nevada near the middle of California and flows in a northwesterly direction through the southern portion of the Central Valley. The alluvial fan of the Kings River separates the San Joaquin from the Tulare River basin.

Human Activities and their Impacts

Land Use. Land use is dominated by forest (and vacant land) in northern basins, and grass, shrubland, and urban uses dominate in southern basins (see Table 35). Overall, the most developed watersheds are the Santa Ana, Russian, and Santa Margarita Rivers. The Santa Ana Watershed encompasses portions of San Bernardino, Los Angeles, Riverside, and Orange counties. About 50% of coastal subbasin of the Santa Ana watershed is dominated by urban land uses and the population density is about 1,500 people per square mile. When steep and unbuildable lands are excluded from this area, then the population density in the watershed is 3,000 people per square mile. However, the most densely populated portion of the basin is near the city of Santa Ana where density reaches 20,000 people per square mile (Burton *et al.* 1998, Belitz *et al.* 2004). The basin is home to nearly 5 million people and the population is projected to increase two-fold in the next 50 years (Burton *et al.* 1998, Belitz *et al.* 2004).

Not only is the Santa Ana watershed the most heavily developed watersheds in the region, the Santa Ana is the most heavily populated study site out of more than 50 assessment sites studied across the nation by the United States Geological Survey under the National Water-Quality Assessment (NAWQA) Program. Water quality and quantity in the basin reflects the influence of the high level of urbanization. For instance, the primary source of baseflow to the river is the treated wastewater effluent; secondary sources--sources that influence peak flows—include stormwater runoff from urban, agricultural, and undeveloped lands (Belitz *et al.* 2004). Concentrations of nitrates and pesticides are elevated within the basin, and were more frequently detected than in other national NAWQA sites (Belitz *et al.* 2004). Belitz *et al.* (2004)found that total nitrogen concentrations commonly exceeded 3 mg/L in the Santa Ana basin. In other NAWQA basins with elevated total nitrogen concentrations across the country, the primary influencing factor was the level of agriculture and the application of manure and pesticides within the basin. In the Santa Ana basin the elevated nitrogen is attributed largely to the wastewater treatment plants, where downstream reaches consistently exceeding 3 mg/L total nitrogen. Samples of total nitrogen taken upstream of the wastewater treatment plants were commonly below 2 mg/L (Belitz *et al.* 2004). Other contaminants detected at high levels included volatile organic compounds (VOCs; including chlorform, which sometimes exceeded water quality standards), pesticides (including diuron, diazinon, carbaryl, chlophyrifos, lindane, malathion, and chlorothalonil), and trace elements (including lead, zinc, arsenic). As a result of the changes, the biological community in the basin is heavily altered (Belitz *et al.* 2004).

Table 35. Land Uses and Population Density in Several Southwest Coast Region (Carter and Resh 2005).

Watershed	Land Use Categories (Percent)				Density (people/mi.^{2})
	Agriculture	Forest	Urban	Other	
Rogue River	6	83	<1	9 grass & shrub	32
Klamath River	6	66	<1	24 grass, shrub, wetland	5
Eel River	2	65	<1	31 grass & shrub	9
Russian River	14	50	3	31 (23 grassland)	162
Sacramento River	15	49	2	30 grass & shrub	61
San Joaquin River	30	27	2	36 grass & shrub	76
Salinas River	13	17	1	65 (49 grassland)	26
Santa Ana River	11	57	32	---	865
Santa Margarita River	12	11	3	71 grass & shrub	135

In many basins, agriculture is the major water user and the major source of water pollution to surface waters. In 1990 nearly 95% of the water diverted from the San Joaquin River was diverted for agriculture, and 1.5% diverted for livestock (Carter and Resh 2005). During the same period, Fresno, Kern, Tulare, and Kings Counties ranked top in the nation for nitrogen fertilizer use. Nitrogen fertilizer use increased 500% and phosphorus use increased 285% in the San Joaquin River basin in a 40 year period (Knatzer and Sheton 1998 *in*) (Carter and Resh 2005). A study conducted by USGS in the mid-1990s on water quality within San Joaquin River basin detected 49 pesticides in the mainstem and three subbasins--22 pesticides were detected in 20% of the samples and concentrations of seven exceeded water quality standards (Dubrovsky *et al.* 1998). Water chemistry in the Salinas River is strongly influence by intensive agriculture—water hardness, alkalinity, nutrients and conductivity are high in areas where agricultural uses predominate.

Mining. Famous for the gold rush of the mid 1800s, California has a long history of mining. In 2004, California ranked top in the nation for nonfuel mineral production with 8.23% of the total production (NMA 2007). Today, gold with silver and iron ore comprise only 1% of the production value. Primary minerals include construction sand and gravel, cement, boron and crushed stone. California is the only state to produce boron, rare-earth metals and asbestos (NMA 2007).

The State contains some 1,500 abandoned mines and roughly 1% is suspected of discharging metal-rich waters in the basins. The Iron Metal Mine in the Sacramento Basin releases more tha 500 kg of copper and more than 350 kg of zinc to the Keswick Reservoir below Shasta Dam, as well as elevated levels of lead (Cain *et al.* 2000) (Cain *et al.* 2000 in)(Carter and Resh 2005). Metal contamination seriously reduces the biological productivity within a basin, can result in fish kills at high levels and at low levels contributes to sublethal effects including reduced feeding, overall activity levels, and growth. The Sacramento Basin and the San Francisco Bay watershed is one of the most heavily impacted basins within the state from mining activities, largely because the basin drains some of the most productive mineral deposits in the region. Methylmercury contamination within San Francisco Bay, the result of 19th century mining practices using mercury to amalgamate gold in the Sierra Nevada Mountains, remains a persistent problem today. Based on sediment cores, we know that pre-mining concentrations were about 5 times lower than concentrations detected within the Bay today (Conaway *et al.* 2003, EPA 2006a).

Hydromodification Projects. Several of the rivers within the area have been modified by dams, water diversions and drainage systems for agriculture and drinking water, and some of the most drastic channelization projects within the nation. In all, there are about 1,400 dams within the State of California, more than 5,000 miles of levees, and more than 140 aqueducts (Mount 1995). While about 75% of the runoff occurs in basins in the northern half of the State, 80% of the water demand is in the southern half of the State. Two water diversion projects meet these demands—the Federal Central Valley Project and the California State Water Project. The Central Valley Project, one of the world's largest water storage and transport systems, has more than 20 reservoirs and delivers about 7 million acre-feet each year to southern California. The State Water Project has 20 major reservoirs and holds nearly 6 million acre-feet of water, delivering about 3 million acre feet. Together these diversions irrigate about 4 million acres of farmland and deliver drinking water to about 22 million residents.

Both the Sacramento River and the San Joaquin River are heavily modified, each with hundreds of dams. The Rogue, Russian, and Santa Ana Rivers each have more than 50 dams, and the Eel, Salinas, and the Klamath Rivers have between 14 and 24 dams. The Santa Margarita, considered one the last free flowing rivers in coastal southern California has 9 dams in its watershed. All major tributaries of the San Joaquin River are impounded at least once and most have multiple dams or diversions. The Stanislaus River, a tributary of the San Joaquin River has over 40 dams. As a result, the hydrograph of the San Joaquin River is seriously altered from its natural state, the temperature regime and sediment transport regime are altered, and such changes have had profound influences on the biological community within the basin—while the modifications generally result in a reduction of suitable habitat for native species, these changes frequently result in a concomitant increase of suitable habitat for nonnative species. The Friant Dam on the San Joaquin River is attributed with the extirpation of spring-run Chinook salmon within the basin, a run once estimated as producing 300,000 to 500,000 fish (Carter and Resh 2005).

Commercial and Recreational Fishing. The region is home to many commercial fisheries. The largest in terms of total landings in 2006 were northern anchovy, Pacific sardine, Chinook salmon, sablefish, Dover sole, Pacific whiting, squid, red sea urchin, and Dungeness crab (CDFG 2007). Red abalone are also harvested off of the shores of California. Illegal poaching of abalone, including endangered white abalone continues to be of concern in the state, with the demand for abalone in local restaurants, seafood markets and international businesses (Daniels and Floren 1998). The first salmon cannery established along the west coast was located in the Sacramento River watershed in 1864 but it only operated for about two years because the sediment from hydraulic mining decimated the runs in the basin (Hittell 1882, and Goode and others, 1884-1887, cited in NRC 1996).

Ballast Water Bioregion NEP-IV (Oregon and Washington Coasts)

This region encompasses Washington, Oregon, Idaho, and includes parts of Nevada, Montana, Wyoming, and British Columbia. The region is ecologically diverse, encompassing northern marine lowland forests, mountain forests, alpine meadows and Northern desert habitat. In this section we focus on three primary areas that characterize the region, the Columbia River Basin and its tributaries, the Puget Sound Region, and the Coastal Drainages north of the Columbia River. The broader ecoregion divisions, as defined by Bailey (1995), and encompassed within this region are the Marine and Marine Mountains Divisions, portions of the Temperate Dessert, and Temperate Steppe and Temperate Steppe Mountains. Puget Sound and the coastal drainages are contained within the Marine Division, while the Columbia River watershed encompasses portions of all five ecoregions.

Columbia River Basin

Natural History. The most notable of all basins within the region is the Columbia River. The largest river in the Pacific Northwest and the fourth largest river in terms of average discharge the United States drains an area over 258,000 square miles (making it the sixth largest in terms of drainage area), the Columbia River Basin includes parts of Washington, Oregon, Nevada, Utah, Idaho, Wyoming, Montana and British Columbia and encompasses 13 terrestrial and three freshwater ecoregions, including arid shrub-steppes, high desert plateaus, temperate mountain forests, and deep gorges (Kammerer 1990, Hinck *et al.* 2004, Stanford and Synder 2005).

Major tributaries include the Snake, Willamette, Salmon, Flathead, and Yakima Rivers; smaller rivers include the Owyhee, Grande Ronde, Clearwater, Spokane, Methow, Cowlitz and the John Day Rivers (see Table 36 for a description of select Columbia River Tributaries). The Snake River is the largest tributary at more than 1,000 miles long; its headwaters originating in Yellowstone National Park, Wyoming. The second largest tributary is the Willamette River in Oregon (Kammerer 1990, Hinck *et al.* 2004). The Willamette River is the 19th largest river in the nation in terms of average annual discharge (Kammerer 1990). The basins drain portions of the Rocky Mountains, the Bitteroot Range, and the Cascade Mountain Range.

The average annual runoff at the mouth of the Columbia River is 265,000 cubic feet per second (cfs)(Kammerer 1990). A saltwater wedge extends 23 miles upstream of the mouth with tidal influences extending up to 146 miles up river (Hinck *et al.* 2004). The climate within the basin is a mix of arid, dry summers, cold winters, and maritime air masses entering from the west. It is not uncommon for air temperatures in the Rocky Mountains to dip below zero in mid-winter, but summer air temperatures can reach more than 100 °F in the middle basin.

Table 36. Select Tributaries of the Columbia River (Carter and Resh 2005)

Watershed	Length (mi. [approx.)	Basin Size (mi^2)	Physiographic Provinces*	Mean Annual Precipitation (inches)	Mean Discharge (cfs).	No. Fish Species (native)	No. Endangered Species
Snake/Salmon River	870	108,495	CU, NR, MR, B/R	14	55,267	39 (19)	5 fish (4 T, 1 E), 6 (1 T, 5 E) snails, 1 plant (T)
Yakima River	214	6,139	CS, CU	7	3,602	50	2 (T)
Willamette River	143	11,478	CS, PB	60	32,384	61 (~31)	5 fish (4 T, 1 E),

* Physiographic Provinces: CU = Columbia-Snake River Plateaus, NR = Northern Rocky Mountains, MR = Middle Rocky Mountains, B/R=Basin & Range, CS = Cascade-Sierra Mountains, PB = Pacific Border

The river and estuary were once home to more than 200 distinct runs of Pacific salmon and steelhead, and represented adaptation to the local environment within a tributary or segment of a river (Stanford and Synder 2005). Salmonids within the basin include Chinook, chum, coho, sockeye salmon, steelhead and redband trout, bull trout, and cutthroat trout. Other fish species within the basin include sturgeon, eulachon, lamprey, and sculpin (Wydoski and Whitney 1979a). According to a review by Stanford *et al.* (2005), the basin contained 65 native fish species and at least 53 nonnative fishes. The most abundant non-native fish is the American shad, which was introduced to the basin in the late 1800s (Wydoski and Whitney 1979a).

Human Activities and their Impacts

Land Use. More than 50% of the United State's portion of the Columbia River Basin is in Federal ownership (most of which occurs in high desert and mountain areas), 39% is in private land ownership (most of which occurs in river valleys and plateaus), and the remainder is divided among tribes, state, and local governments (Hinck *et al.* 2004). See Table 37 for a summary of land uses and population densities in several subbasins within the Columbia River watershed (Stanford and Synder 2005).

Table 37. Land Uses and Population Density in Select Tributaries of the Columbia River (Stanford and Synder 2005)

Watershed	Land Use Categories (Percent)				Density (people/mi.^{2})
	Agriculture	Forest	Urban	Other	
Snake/Salmon River	30	10-15	1	54 scrub/rangeland/barren	39
Yakima River	16	36	1	47 shrub	80
Willamette River	19	68	5	--	171

The interior Columbia Basin has been altered substantially by humans causing dramatic changes and declines in many native fish populations. In general the basin supports a variety of mixed uses. Predominant human uses include logging, agriculture, ranching, hydroelectric power generation, mining, fishing and a variety of recreational activities, and urban uses.

The decline of salmon runs in the Columbia is attributed to loss of habitat, blocked migratory corridors, altered river flows and pollution, over harvest, and competition from hatchery fish. Critical ecological connectivity (mainstem to tributaries and riparian floodplains) has been disconnected by dams and associated activities such as floodplain deforestation and urbanization. The most productive floodplains of the watershed are either flooded by hydropower dams or dewatered by irrigation diversions. Portions of this basin are also subject to impacts from cattle grazing and irrigation withdrawals. In the Yakima River 72 stream and river segments are listed as impaired by the Washington Department of Ecology and 83% exceed temperature standards. In the Willamette River riparian vegetation was greatly reduced by land conversion. By 1990 only 37% of the riparian area within 120 m was forested, 30% was agricultural fields and 16% was urban or suburban lands. In the Flathead River aquatic invasive plants such as pondweed, hornwort, watermilfoil, waterweed, cattail and duckweed grow in the floodplain wetlands and shallow lakes and in the Yakima River non-native grasses and other plant are commonly found along the lower reaches of the river (Stanford and Synder 2005).

Agriculture and Ranching. Roughly 6% of the annual flow from the Columbia River is diverted for the irrigation of 7.3 million acres of croplands within the basin. The vast majority of these agricultural lands are located along the lower Columbia River, the Willamette, Yakima, Hood, and Snake Rivers, and the Columbia Plateau (Hinck *et al.* 2004). The Yakima River Basin is one of the most agriculturally productive areas in the United States (Fuhrer *et al.* 2004).

Croplands within the Yakima Basin account for about 16% of the total basin area of which 77% is irrigated.

Agriculture and ranching increased steadily but slowly within the Columbia River basin from the mid to late 1800. By the early 1900s, agricultural opportunities began increasing at a much more rapid pace with creation of more irrigation canals and the passage of the Reclamation Act of 1902 (NRC 2004). Today, agriculture represents the largest water use within the basin. More than 105,000 acre feet per day (more than 90 percent) is used for agricultural purposes. Agriculture, ranching, and the related services employ more than nine times the national average (19% of the households within the basin; (NRC 2004)).

Ranching practices have led to increased soil erosion and sediment loads within adjacent tributaries, the worst of these effects may have occurred in the late 1800s and early 1900s with deliberate burning to increase grass production (NRC 2004). Several measures are in use to reduce the impacts of grazing including restricting grazing in degraded areas, reduced grazing allotments, and lower stocking rates. Today agricultural impacts to water quality within the basin are second to large scale influences of hydromodification projects for both power generation and irrigation. Water quality impacts from agricultural activities include alteration of the natural temperature regime, and insecticide and herbicide contamination, and increased suspended sediments.

The US Geological Survey has a number of fixed water quality sampling sites throughout various tributaries of the Columbia River, many of which have been in place for decades. Water volumes, crop rotation patterns, crop-type, and location of within the basin are some of the variables that influence the distribution and frequency of pesticides within a tributary. Detection frequencies for a particular pesticide can vary widely. One study conducted by the US Geological Survey between May 1999 and January 2000, detected 25 pesticide compounds (Ebbert and Embry 2001). Another study detected at least two pesticides or their breakdown products in 91% of the samples collected, with the median number of chemicals being eight, and the maximum was 26. The herbicide 2,4-D occurred most often in the mixtures, along with azinphos-methyl, the most heavily applied pesticide, and atrazine, one of the most mobile pesticides in water (Fuhrer *et al.* 2004). However, the most frequently detected pesticides in the Yakima River Basin are total DDT, as well as its breakdown products DDE and DDD, and dieldrin (Johnson and Newman 1983, Joy 2002, Joy and Madrone 2002, Fuhrer *et al.* 2004). In addition to current use-chemicals these legacy chemicals continue to pose a serious problem to water quality and fish communities despite their cancellation in the 1970s and 1980s (Hinck *et al.* 2004).

Fish and macroinvertebrate communities exhibit an almost linear decline in condition as the level of agriculture intensity increases within a basin (Cuffney *et al.* 1997, Fuhrer *et al.* 2004). A study conducted in the late 1990s examining 11 species of fish, including anadromous and resident fish collected throughout the basin for a suite of 132 contaminants, which included 26 pesticides revealed organochlorines, specifically hexachlorobenzene, chlordane and related compounds, and DDT and its metabolites, were the most frequently detected pesticides within fish tissues (Hinck *et al.* 2004).

Urban and Industrial Development. The largest urban area in the basin is the greater Portland metropolitan area, located at the mouth of the river. Portland's population exceeds 500,000 people, whereas the next largest cities, Spokane, Salem, Eugene, and Boise, have more than 100,000 people (Hinck *et al.* 2004). Overall, however the population within the basin is one-third the average, and while the basin covers about 8% of United States' land, only about 1.2% of the United States population lives within the basin (Hinck *et al.* 2004).

Discharges from sewage treatment plants, paper manufacturing, and chemical and metal production represent the top three permitted sources of contaminants within the lower basin according to discharge volumes and concentrations (Rosetta and Borys 1996). According to Rosetta and Borys (1996) based on their review of 1993 data, 52% of the point source waste water discharge volume is from sewage treatment plants, 39% from paper and allied products, 5% from chemical and allied products, and 3% from primary metals. However, suspended sediment loading is predominantly from point sources from the paper and allied products industry (71%), while 26% comes from sewage treatment plants and 1% is from the chemical and allied products industry. Non-point source discharges (urban stormwater runoff) account for more of the total pollutant loading to the lower basin for most organics and over half of the metals. Although rural non-point sources contributions were not calculated, Rosetta and Borys (1996) surmised that in some areas and for some contaminants rural areas may contribute a large portion of the load; this is particularly the case for pesticide contamination in the upper river basin where agriculture is the predominant land use.

A study conducted in the late 1990s examining 11 species of fish, including anadromous and resident fish collected throughout the basin for a suite of 132 contaminants, which included 51 semi-volatile chemicals, 26 pesticides, 18 metals, seven PCBs, 20 dioxins, and 10 furans revealed PCBs, metals, chlorinated dioxins and furans (products of wood pulp bleaching operations) and other contaminants within fish tissues—white sturgeon tissues contained the greatest concentrations of chlorinated dioxins and furans (Hinck *et al.* 2004).

Hydromodification Projects. More than 400 dams exist in the basin ranging from mega dams that store large amounts of water to small diversion dams for irrigation. Every major tributary of the Columbia except the Salmon River is totally or partially regulated by dams and diversions. More than 150 dams are major hydroelectric projects of which 18 dams are located on mainstem Columbia River and its major tributary, the Snake River. The Federal Columbia River Power System encompasses the operations of 14 major dams and reservoirs on the Columbia and Snake Rivers, operated as a coordinated system. The Army Corps of Engineers operates nine of 10 major Federal projects on the Columbia and Snake Rivers, and Dworshak, Libby and Albeni Falls dams. The Bureau of Reclamation operates Grand Coulee and Hungry Horse dams. These Federal projects are a major source of power in the region, and provide flood control, navigation, recreation, fish and wildlife, municipal and industrial water supply, and irrigation benefits.

The Bureau of Reclamation has operated irrigation projects within the basin since the 1904. The irrigation system delivers water to about 2.9 million acres of agricultural lands; 1.1 million acres of land are irrigated using water delivered by two structures, the Columbia River Project (Grand Coulee Dam) and the Yakima Project. Grand Coulee Dam delivers water for the irrigation of over 670,000 acres of crop lands and the Yakima Project delivers water to nearly 500,000 acres of crop lands (BOR 2007).

The Bonneville Power Administration, an agency of the US Department of Energy, wholesales electric power produced at 31 Federal dams (67% of its production) and non-hydropower facilities in the Columbia-Snake Basin, selling about half the electric power consumed in the Pacific Northwest. The Federal dams were developed over a 37-year period starting in 1938 with Bonneville Dam and Grand Coulee in 1941, and ending with construction of Libby Dam in 1973 and Lower Granite Dam in 1975.

Development of the Pacific Northwest regional hydroelectric power system, dating to the early twentieth century, has had profound effects on the ecosystems of the Columbia River Basin (ISG 1996). These effects have been especially adverse to the survival of anadromous salmonids. The construction of the Federal power system modified migratory habitat of adult and juvenile salmonids, and in many cases presented a complete barrier to habitat access. Both upstream and downstream migrating fish are impeded by the dams, and a substantial number of juvenile salmonids are killed and injured during downstream migrations. Physical injury and direct mortality occurs as juveniles pass through turbines, bypasses, and spillways. Indirect effects of passage through all routes may include disorientation, stress, delays in passage, and exposure to high concentrations of dissolved gases, warm water, and increased predation. Dams have also flooded historical spawning and rearing habitat with the creation of massive water storage reservoirs. More than 55% of the Columbia River Basin that was accessible to salmon and steelhead before 1939 has been blocked by large dams (NWPPC 1986). Construction of Grand Coulee Dam blocked 1,000 miles of habitat from migrating salmon and steelhead (Wydoski and Whitney 1979a). The mainstem habitats of the lower Columbia and Willamette Rivers have been reduced primarily to a single channel. As a result, floodplain area is reduced, off-channel habitat features have been eliminated or disconnected from the main channel, and the amount of large woody debris in the mainstem has been reduced. Remaining areas are affected by flow fluctuations associated with reservoir management for power generation, flood control and irrigation. Overbank flow events, important to habitat diversity, have become rare as a result of controlling peak flows and associated revetments. Consequently, the dynamics of estuary has changed substantially.

Artificial Propagation. There are several artificial propagation programs for salmon production within the Columbia River Basin, many of which were instituted under Federal law to ameliorate the effects of lost natural production of salmon within the basin from the dams on fishing. The hatcheries are operated by Federal, state, and tribal managers. For more than 100 years, hatcheries in the Pacific Northwest have been used to produce fish for harvest and replace natural production lost to dam construction, and have only minimally been used to protect and rebuild naturally produced salmonid population (e.g., Redfish Lake sockeye salmon). In 1987, 95% of the coho salmon, 70% of the spring Chinook salmon, 80% of the summer Chinook salmon, 50% of the fall Chinook salmon, and 70% of the steelhead returning to the Columbia River Basin originated in hatcheries (CBFWA 1990). More recent estimates suggest that almost half of the total number of smolts produced in the basin come from hatcheries (Mann *et al.* 2005).

The impact of artificial propagation on the total production of Pacific salmon and steelhead has been extensive (Hard *et al.* 1992). Hatchery practices, among other factors, are a contributing factor to the 90% reduction in natural coho salmon runs in the lower Columbia River of the past 30 years (Flagg *et al.* 1995). Past hatchery and stocking practices have resulted in the transplantation of salmon and steelhead from nonnative basins, and the impacts of these practices are largely unknown. Adverse effects of these practices likely included: the loss of genetic variability within and among populations (Busack 1990 and Riggs 1990 cited in Hard *et al.* 1992) (Reisenbichler 1997), disease transfer; increased competition for food, habitat, or mates; increased predation; altered migration; and displacement of natural fish (Steward and Bjornn 1990 *cited in* Hard *et al.* 1992) (Hard *et al.* 1992, Fresh 1997); and species with extended freshwater residence are likely to face higher risk of domestication, predation, or altered migration than are species that spend only a brief time in fresh water (Hard *et al.* 1992) to name a few. Nonetheless, artificial propagation also may contribute to the conservation of listed salmon and steelhead although it is unclear whether or how much artificial propagation during the recovery process will compromise the distinctiveness of natural population (Hard *et al.* 1992).

Currently, NMFS is working on hatchery reform project in the Columbia River Basin, which will include a collaborative review of how harvest and hatcheries—particularly federally-funded hatcheries— are affecting the recovery of listed salmon and steelhead in the Basin. Eventually, the project team would create a management approach that allows tribal, state and Federal managers to effectively manage Columbia River Basin hatcheries to meet conservation and harvest goals consistent with their respective legal responsibilities. This effort was mandated by Congress in 2005, and is currently in its early stages.

Mining. Most of the mining in the basin is focused on minerals such as phosphate, limestone, dolomite, perlite, or metals such as gold, silver, copper, iron and zinc. Mining in the region is conducted in a variety of methods and places within the basin. Alluvial or glacial deposits are often mined for gold or aggregate, and ores are often excavated from the hard bedrocks of the Idaho batholiths. Eleven percent of the nation's output of gold has come from mining operations in Washington, Montana, and Idaho, and more than half of the nation's silver output has come from a few select silver deposits with 30% coming from two deposits located in the Columbia River Basin (the Clark Fork River and Coeur d'Alene deposits; (Hinck *et al.* 2004, Butterman and Hilliard 2005). According to Wydoski and Whitney (1979a) one of the largest mines in the region, located near Lake Chelan, once produced up to 2,000 tons of copper-zinc ore with gold and silver on a daily basis. Most of the phosphate mining within the basin occurs within the headwaters of the Snake River, but the overall output from these deposits accounts for 12% of the United States production of phosphate (Hinck *et al.* 2004).

Many of the streams and river reaches in the basin are impaired from mining and several abandoned and former mining sites are designated as superfund cleanup areas (Stanford and Synder 2005, EPA 2007). According to the US Bureau of Mines, there are about 14,000 inactive or abandoned mines within the Columbia River Basin of which nearly 200 pose a potential hazard to the environment (Quigley and Graham 1997). Contaminants that have been detected in the water include lead and other trace metals. Mining of copper, cadmium, lead, manganese, and zinc in the upper Clark Fork River have contributed wastes to this basin since 1880 (Woodward *et al.* 1994). Benthic macroinvertebrates and fish within the basin have bioaccumulated metals—the exposure and bioaccumulation of these metals in native fishes in the basin are suspected of reducing their survival and growth (Farag *et al.* 1994, Woodward *et al.* 1994). In the Clark River, several fish kills have occurred since 1984 and are attributed to contamination from trace metals such as cadmium, copper, lead and zinc (Hinck *et al.* 2004).

Commercial, Recreational, and Subsistence Fishing. Archeological records indicate that indigenous people caught salmon in the Columbia River more than 7,000 years ago. One of the most well known tribal fishing sites within the basin was located near Celilo Falls, an area in the lower river that has been occupied by Dalles Dam since 1957. Salmon fishing increased with better fishing methods and preservation techniques, such as drying and smoking, such that harvest substantially increased in the mid-1800s with canning techniques. Harvest techniques also changed over time, from early use of hand-held spears and dip nets, to river boats that used seines and gill-nets, eventually, transitioning to large ocean-going vessels with trolling gear and nets and the harvest of Columbia River salmon and steelhead off the waters of the entire west coast, from California to Alaska (Mann *et al.* 2005).

During the mid 1800s, an estimated 10 to 16 million adult salmon of all species entered the Columbia River each year. Large harvests of returning adult salmon during the late 1800s ranging from 20 million to 40 million pounds of salmon and steelhead annually significantly reduced population productivity (Mann *et al.* 2005). The largest harvest of Chinook salmon ever recorded occurred in 1883 when Columbia River canneries processed 43 million pounds of salmon (Lichatowich 1999). Commercial landings declined steadily from the 1920s to a low in 1993, when just over one million pounds were harvested (Mann *et al.* 2005).

Harvested and spawning adults reached 2.8 million in the early 2000s, of which almost half are hatchery produced (Mann *et al.* 2005). Most of the fish caught in the river are steelhead and spring/summer Chinook salmon, while ocean harvest consists largely of coho and fall Chinook salmon. Most ocean catches are made north of Cape Falcon, Oregon. Over the past five years, the number of spring and fall salmon commercially harvested in tribal fisheries has averaged between 25,000 and 110,000 fish (Mann 2004). Recreational catch in both ocean and in-river fisheries varies around 140,000 to 150,000 fish (Mann *et al.* 2005).

Puget Sound Region

Natural History. The Puget Sound watershed defined by the crest lines of the Olympia Mountain Range (and the Olympic Peninsula) to the west and the Cascade Mountain Range to the east. The Olympic Mountains reach heights of about 8,000 feet above sea level, and are extremely rugged and steeply peaked with abrupt descents into the Puget Lowland. The Cascade Mountains on the east range in heights of 4-8,000 feet above sea level with the highest peak, Mount Rainer towering over the region at 14,410 feet above sea level. As the second largest estuary in the United States, Puget Sound has about 1330 miles of shoreline, extends from the mouth of the Strait of Juan de Fuca east, including the San Juan Islands and south to Olympia, and is fed by more than 10,000 rivers and streams.

Puget Sound is generally divided into four major geographic marine basins: Hood Canal, South Sound, Whidbey Basin, and the Main Basin. The Main Basin has been further subdivided into two sub-basins: Admiralty Inlet and Central Basin. Each of the above basins forms a depression on the sea floor in which a shallower ledge or sill separates the relatively deep water from the adjacent basin. The waters of Puget Sound function as a partially mixed, two-layer system, with relatively fresh water flowing seaward at the surface and salty oceanic water entering at depth. The main ledge of Puget Sound is located at the north end of Admiralty Inlet where the water shoals to a depth of about 200 feet at its shallowest point (King County 2001). The deepest point in Puget Sound is found in the Central Basin and is over 920 feet. Approximately 43% of the Puget Sound's tideland is located in the Whidbey Island Basin. This reflects the large influence of the Skagit River, which is the largest river in the Puget Sound system and whose sediments are responsible for the extensive mudflats and tidelands of Skagit Bay.

Habitat types that occur within the nearshore environment include eelgrass meadows, kelp forest, mud flats, tidal marshes, subestuaries (tidally influenced portions of river and stream mouths), sand spits, beaches and backshore, banks and bluffs, and marine riparian vegetation. These habitats provide critical functions such as primary food production, support habitat for invertebrates and juvenile and adult fishes, and provide foraging and refuge opportunities for birds and other wildlife.

The Puget Sound ecoregion is a glaciated area consisting of glacial till, glacial outwash and lacustrine deposits with high quality limestone is found in the San Juan Islands (Wydoski and Whitney 1979a). Relief in the valley is moderate with elevation ranging from sea level to about 1300 feet. Geology in the region consists of mostly Tertiary sedimentary bedrock formations.

The land and vegetation surrounding Puget Sound waters is classified as Puget Lowland Forest and occupies the depression or valley between the Olympic Peninsula on the west and the Cascade Mountains on the east (Franklin and Dyrness 1973). The alpine zone is expressly devoid of trees. Vegetation changes abruptly along the mountain slopes and across minimal horizontal distances as a result of steep topography, soil, and microclimate (sun exposure, temperature, and precipitation). Dominant vegetation types include from the Puget lowland region – the lowland forest, the mid-montane forest of Pacific silver fir (*Abies amabilis*) with Alaska yellow cedar (*Chamaecyparis nootkatensis*); the subalpine forest of mountain hemlock (*Tsuga mertensiana*)with subalpine fir (*Abies lasiocarpa*) and Alaska yellow cedar; and the alpine tundra or meadow above the treeline (Kruckeberg 1991).

The Puget Sound region has a Mediterranean-like climate, with warm, dry summers, and mild wet winters (Franklin and Dyrness 1973). Annual precipitation varies from 28-35 inches, and falls predominantly as rain in lowland areas. Annual snowpack in the mountain ranges is often high—although the elevation of the Olympia Mountains is not as high as that of the Cascade Mountain Range, abundant accumulation occurs, such that it will sometimes persist throughout much of the summer months. Average annual rainfall in the north Cascades at Mount Baker Lodge is about 110 inches, and at Paradise Station at Mount Rainer is about 105 inches, while average annual snowfall is 550 inches and 582 inches respectively--sometimes reaching more than 1,000 inches on Mount Rainer (Wydoski and Whitney 1979a, Kruckeberg 1991).

Major rivers draining to Puget Sound from the Cascade Mountains include the Skagit River, the Snohomish River, the Nooksack River, the Puyallup/Green River, and the Lake Washington/Cedar River watershed. Major rivers from the Olympic Mountains include the

Hamma Hamma, the Duckabush, the Quilcene, and the Skokomish Rivers. Numerous other smaller rivers drain to the Sound, many of which are significant producers of salmonids despite their small size.

The Puget Sound basin is home to: more than 200 fish species, representing more than 50 families; more than 140 mammals, of which less than a third are marine mammals. Salmonids within the region include coho salmon, Chinook salmon, sockeye salmon and kokanee, chum salmon, pink salmon, steelhead and rainbow trout, coastal cutthroat trout, bull trout, and Dolly Varden (Wydoski and Whitney 1979a, Kruckeberg 1991). Important commercial fishes include the five Pacific salmon species and several rockfish species. A number of introduced species occur within the region including brown trout, brook trout, Atlantic salmon, bass, tunicates (sea squirts), and a saltmarsh grass (spartina). Estimates suggest that more than 90 species have been intentionally or accidentally introduced in the region (Ruckelshaus and McClure 2007). At present over 40 species in the region are listed as threatened and endangered under the ESA.

Human Activities and the Impacts

Land Use. Land use in the Puget Sound lowland is composed of agricultural areas (including forests for timber production), urban areas (industrial and residential use), and rural areas (low density residential with some agricultural activity). In the 1930s, all of Western Washington contained about 15.5 million acres of "harvestable" forest land and by 2004 the total acreage was nearly half that surveyed more than 70 years earlier (PSAT 2007a). Forest cover in Puget Sound alone was about 5.4 million acres in the early 1990s and about a decade later the region had lost another 200,000 acres of forest cover with some watersheds losing more than half the total forested acreage. The most intensive loss of forest cover has occurred in the State's Urban Growth Boundary, which encompasses specific parts of the Puget Lowland; in this area forest cover declined by 11.1% between 1991 and 1999 (Ruckelshaus and McClure 2007). Projected land cover changes (Ruckelshaus and McClure 2007) indicate that trends are likely to continue over the next several decades with population changes—coniferous forests are projected to decline at an alarming rate as urban uses increase.

The Puget Sound Lowland contains the most densely populated area of Washington. The regional population in 2003 was an estimated 3.8 million people, with 86% residing in King, Pierce and Snohomish Counties (Snohomish, Cedar-Sammamish Basin, Green-Duwamish, and Puyallup River watersheds), and the area is expected to attract four to six million new human residents in the next 20 years (Ruckelshaus and McClure 2007).

According to the State of the Sound report (PSAT 2007a) in 2001, impervious surfaces covered 3.3% of the region, with 7.3% of lowland areas (below 1,000 feet elevation) covered by impervious surfaces. In one decade, 1991 – 2001 impervious surfaces increased 10.4% region wide. The Snohomish River watershed, one of the fastest growing in the region, increased 15.7% in the same period.

Much of the region's estuarine wetland losses have been heavily modified, primarily from agricultural land conversion and urban development (NRC 1996b). Although most estuarine wetland losses result from conversions to agricultural land by ditching, draining, or diking, these wetlands are also experiencing increasing effects from industrial and urban causes.

The most extreme case of river delta conversion is observed in the Duwamish Waterway in Seattle. As early as the mid-1800s, settlers in the region began discussing the need for a ship canal that linked Lake Washington directly with Puget Sound. After several private and smaller attempts, by the early 1900s locks were built achieving this engineering feat. The resultant outcome was that the Black River, which formerly drained Lake Washington to the Green and White Rivers (at their confluence, these rivers formed the Duwamish River), dried up. The lower White River, which historically migrated sporadically between the Puyallup and the Green/Duwamish basins, was permanently diverted into the Puyallup River basin in 1914 with the construction of concrete diversion at river mile 8.5, resulting in a permanent increase of the Puyallup River flows by about 50% and a doubling of the drainage area (Kerwin 1999). The Cedar River, on the other hand was permanently diverted to Lake Washington. The oxbow in the lower Duwamish River was lost with the lower river dredging in the early 1900s reducing the lower nine miles of the river to 5 miles in length. Overtime the Waterway has been heavily armored and diked, result in the loss of all tidal swamps, 98% of the tidal forests, marshes, shallows and flats and 80% of the riparian shoreline (Blomberg *et al.* 1988).

By 1980, an estimated 27,180 acres of intertidal or shore wetlands had been lost at eleven deltas in Puget Sound (Bortleson *et al.* 1980). Tidal wetlands in Puget Sound amount to about 17-19% of their historical extent (Collins and Sheikh 2005). Coastal marshes close to seaports and population centers have been especially vulnerable to conversion with losses of 50-90% common for individual estuaries.

More than 100 years of industrial pollution and urban development have affected water quality and sediments in Puget Sound. Many different kinds of activities and substances release contamination into Puget Sound and the contributing waters. Positive changes in water quality in the region, however, are also evident. One of the most notable improvements was the elimination of sewage effluent to Lake Washington in the mid 1960s, which significantly reduced problems within the lake from phosphorus pollution and triggered a concomitant reduction in the cyanobacteria (Ruckelshaus and McClure 2007).

Even so, as the population and industry has risen in the region a number of new and legacy pollutants are of concern. According to the State of the Sound Report (PSAT 2007a) in 2004, more than 1,400 fresh and marine waters in the region were listed as “impaired.” Almost two-thirds of these water bodies were listed as impaired due to contaminants, such as toxics, pathogens, and low dissolved oxygen or high temperatures, and less than one-third had established cleanup plans; more than 5,000 acres of submerged lands (primarily in urban areas; 1% of the study area) are contaminated with high levels of toxic substances, including polybrominated diphenyl ethers (PBDEs—flame retardants), and roughly one-third (180,000 acres) the submerged lands within Puget Sound are considered moderately contaminated. PBDEs biomagnified in the food chain, and in the past 20 years the body burden in harbor seals has increased dramatically from 50 ppb to more than 1,000 ppb. Primary pollutants of concern in Puget Sound include heavy metals, organic compounds, PAHs, PCBs, dioxins, furans, DDT, phthalates, and PBDEs.

Areas of highest concern in Puget Sound are Southern Hood Canal, Budd Inlet, Penn Cove, Commencement Bay, Elliott Bay, Possession Sound, Saratoga Passage, and Sinclair Inlet (DOE 2002). Hypoxic dissolved oxygen concentration (<3 mg/L) were found at several (11 out of 54) stations. Dissolved oxygen concentrations less than 3 mg/L were measured in Hood Canal, Penn Cove, Saratoga Passage, Bellingham Bay, Discovery Bay, Elliott Bay, Strait of Georgia and West Point. Conditions in South Hood Canal were especially severe, with low DO concentration (<5 mg/L) evident year-round. Penn Cove also exhibited re-occurring hypoxia. Low DO was found at 18 other stations, including Saratoga Passage, Discovery Bay, Bellingham Bay, Elliott Bay, Budd Inlet, and Commencement Bay.

In 1989 the Washington State Department of Ecology (DOE) began a program to monitor marine sediment conditions called the Puget Sound Assessment and Monitoring Program (PSAMP). The PSAMP is a multi-agency partnership administered by the Puget Sound Action Team. From 1989-1995 the Marine Sediment Monitoring Program was implemented to characterize baseline sediment quality conditions and trends throughout the Greater Puget Sound area. This was the first large scale evaluation of Puget Sound sediment quality at ambient (i.e. away from point sources of contamination) stations through the Sound. Eighty-six stations were established throughout Puget Sound, Hood Canal, the Strait of Georgia, and the Strait of Juan de Fuca. Stations were grouped in two categories: core stations sampled annually, and rotating stations sampled once every three years alternating between North, Central and South Puget Sound regions. At each station, replicate sediment samples were collected for the analysis of chemical contaminants, sediment variables, and benthic community structure.

Overall, contaminant concentrations at monitoring stations were generally low and below state sediment quality standards. Metals and semi-volatile organic compounds were most frequently detected. The highest metal and organic contamination was found in locations associated with urban and industrial centers. Low metal concentrations were also detected in some rural areas and in deep depositional environments. Contaminant concentrations occasionally exceeded state regulatory sediment quality standards. However, there was not a consistent pattern across years. An exception was mercury in Sinclair Inlet and Dyes Inlet, with concentrations above standards for each of the seven years monitored.

By 2000, annual monitoring of sediments at ten historical PSAMP stations showed mixed trends in recent years for some chemicals found in sediments (DOE 2005). Less than one third (32 percent) of almost 13,000 chemical measurements made were detected during testing. Those detected most often exceeded sediment quality guidelines in urban embayments: Sinclair Inlet (mercury), Thea Foss Waterway (PAHs).

In general, metals concentrations in 2000 were lower than in 1989 thru 1996 more often than they were higher, while the opposite was true of PAHs (DOE 2005). At the Port Gardner and Inner Budd Inlet station, concentrations of a number of priority pollutant and metals also decreased significantly. Individual PAH levels decreased at the Point Pully station, but increased significantly at the Bellingham Bay, Port Gardner, and East Anderson Island stations. Total HPAH and total PAH levels increased significantly at the Strait of Georgia, Bellingham Bay, East Anderson Island, and Budd Inlet stations. These changes may reflect changes in anthropogenic input of contaminants to the estuarine system over this 12-year study period. Also, changes in grain size and benthic infaunal community composition seen at the Strait of Georgia station were probably linked to increased precipitation and subsequent increased flow and sediment loading from the Fraser River in 1996 and 1997.

From 1997 to 1999, sediments were collected throughout Puget Sound as part of a joint monitoring program conducted by the DOE and NOAA (DOE 2003). Analyses were performed to quantify concentrations of potentially toxic chemicals, responses in laboratory toxicity tests, and the structure of benthic infauna communities in sediments.

Degraded conditions, as indicated by a combination of relative high chemical concentrations, statistically significant responses in one or more tests of toxicity, and adversely altered benthos, occurred in samples that represented about 1% of the total area (5,700 acres) (DOE 2003). These conditions occurred in samples collected within urbanized bays and industrial waterways, especially near the urban centers of Everett, Seattle, Tacoma, and Bremerton, where degraded conditions had been reported in previous studies. Sediments with high quality (as indicated by no elevated chemical concentrations, no significant responses in the toxicity tests, and the presence of abundant and diverse infauna and or pollution sensitive taxa) occurred in samples that represented a majority, 68% of the total study area (400,000 acres). Sediments in which results of the three kinds of analyses were not in agreement were classified as intermediate in quality and represented about 31% of the total area (179,000 acres).

Although the highly degraded sediments comprise a small percentage of Puget Sound’s area these hot spots upload pollution into the food web, and the resulting damage to the ecological health and function of the Puget Sound ecosystem may be much greater than the small area suggest.

Researchers detected arsenic, copper, lead, and mercury throughout the Sound. They found cadmium at 59% of the stations and tributulin, an antifouling chemical found in ship hull paint, at 50% of the stations. PAHs were common while phthlalate esters, PCBs, DDTs and dibenzofurans appeared at fewer stations (PSAT 2004). Degraded sediments were most prevalent in the Whidbey Basin and Central Sound regions (Everett Harbor, Elliott Bay, Commencement Bay). A higher degree of degradation in critical nearshore habitat may disproportionately affect important fish, shellfish and aquatic plant species (DOE 1997-2003 posters).

The USGS assessed water quality of streams, rivers and groundwater in the Puget Sound Basin as part of the National Water-Quality Assessment (NAWQA) Program between 1996 and 1998. This assessment focused on the quality of surface and ground waters and biological indicators such as fish status, algal status and invertebrate status in relation to land use. A widespread detection of pesticide compounds was observed in surface waters of the Puget Sound Basin (Bortleson and Ebbert 2000). Slightly more than half of the pesticide compounds (26 of 47 analyzed) were detected. The study found that large rivers in the Puget Sound Basin were more likely to meet Federal and state guidelines than were small streams (Ebbert *et al.* 2000). A total of 74 manmade organic chemicals were detected in streams and rivers, with different mixtures of chemicals linked to agricultural and urban settings including atrazine, prometon, simazine and tebuthiuron, carbaryl, diazinon, and malathion (Bortleson and Ebbert 2000). Commonly detected volatile organic compound in the agricultural land-use study area was associated with the application of fumigants to soils prior to planting (Ebbert *et al.* 2000). The average concentration of total nitrogen in small streams draining agricultural lands was twice the concentration in streams draining urban areas and over 40 times the concentration in streams draining undeveloped areas (Ebbert *et al.* 2000). The study concluded that contaminants in runoff from urban and agricultural land surfaces were major influences on the water quality of streams and rivers (Ebbert *et al.* 2000), and according to the State of the Sound report water quality impacts from stormwater and wastewater runoff is a major limiting factor in the recovery of salmon and bull trout (Psat 2007b).

Hydromodification Projects. More than 20 dams occur within the region's rivers and overlap with the distribution of salmonids, and a number of basins contain water withdrawal projects or small impoundments that can impede migrating salmon. The resultant impact of these and land use changes (forest cover loss and impervious surface increases) has been a significant modification in the seasonal flow patterns of area rivers and streams, and the volume and quality of water delivered to Puget Sound waters. Several rivers have been hydromodified by other means including levees and revetments, and bank hardening for erosion control, and agriculture uses. The first dike built in the Skagit River delta was built in 1863 for agricultural development (Ruckelshaus and McClure 2007), other basins like the Snohomish River are diked and have active drainage systems to drain water after high flows that top the dikes. Dams were also built on the Cedar, Nisqually, White, Elwha, Skokomish, Skagit and several other rivers in the early 1900s to supply urban areas with water, prevent downstream flooding and allow for floodplain activities (like agriculture or development), and to power local timber mills (Ruckelshaus and McClure 2007).

In the next couple of years, however a highly publicized and long discussed dam removal project is expected to begin in the Elwha River. The removal of two dams in the Elwha River, a short but formerly very productive salmon river, is expected to open up more than 70 miles of high quality salmon habitat (Wunderlich *et al.* 1994). Estimates suggestion that nearly 400,000 salmon could begin using the basin within 30 years after the dams are removed (Psat 2007b).

About 800 miles of Puget Sound's shorelines are hardened or dredged (Psat 2007b). The area most intensely modified is the urban corridor (eastern shores of Puget Sound0 from Mukilteo to Tacoma); here nearly 80% has been altered, mostly from shoreline armoring associated with the Burlington Northern Railroad tracks (Ruckelshaus and McClure 2007). Levee development within the rivers and their deltas has isolated significant portions of former floodplain habitat that was historically used by salmon and trout during rising flood waters.

Mining. Mining has a long history in the State of Washington, and in 2004 the state was ranked 13th nationally in total nonfuel mineral production value and 17th in coal production (Palmisano *et al.* 1993a, NMA 2007). Metal mining for all metals (e.g., zinc, copper, lead, silver, and gold) peaked in the State between 1940 and 1970 (Palmisano *et al.* 1993a). Today, construction sand and gravel, Portland cement and crushed stone are the predominant materials mined. Where sand and gravel is mined from riverbeds (gravel bars and floodplains) it may result in changes in channel elevations and patterns, instream sediment loads, and seriously alter instream habitat. In some cases, instream or floodplain mining has resulted in large scale river avulsions. The effect of mining in a stream or reach depends upon he rate of harvest and the natural rate of replenishment, as well as flood and precipitation conditions during or after the mining operations.

Commercial and Recreational Fishing. Most of the commercial landings in the region are groundfish, Dungeness crab, shrimp, and salmon. Many of the same species are sought by Tribal fisheries, and by charter, and recreational anglers. Nets and trolling are used in commercial and Tribal fisheries, whereas recreational anglers typically use hook and line, and may fish from boat, river bank, and docks. Entanglement of marine mammals in fishing gear is not uncommon and can lead to mortality or serious injury.

Oregon/Washington Coastal Drainages

Natural History. This region encompasses drainages originating in the Klamath Mountains, the Oregon Coast Mountains and the Olympic Mountains--the Coast Range ecoregion where elevations range from sea level to about 4,000 feet. More than 15 watersheds drain the region's steep slopes including the Umpqua, Alsea, Yaquina, Nehalem, Chehalis, Quillayute, Queets, and Hoh Rivers. Numerous other small to moderately sized streams dot the coastline. Many of the basins in this region are relatively small—the Umpqua River drains a basin of 4,685 sq. miles and is a little over 110 miles long and the Nehalem River drains a basin of 855 sq. miles and is almost 120 miles long—yet represent some of the most biologically diverse basins in the Pacific Northwest (Johnson 1999(Kagan *et al.* 1999, Carter and Resh 2005).

The region is part of a coastal, temperate rainforest system, and is characterized by moderate maritime climate marked by long wet seasons with short dry seasons and mild to cool year-round temperatures. Average annual precipitation ranges from about 60 inches to more than 180 inches, much of which falls as rain, and supports a rich temperate forest. Vegetation is characterized by giant coniferous forests of Sitka spruce, western hemlock, Douglas fir, western red cedar, and red alder and black cottonwood

The Oregon Coast supports a unique coastal sand dune system. The sand dunes were largely created by the sand deposited from the coastal rivers, in particular the Umpqua and Columbia Rivers. North, steep headlands and cliffs are separated by stretches of flat coastal plain and large estuaries. Significant estuaries in the region (outside of the Columbia River estuary) include Coos Bay, Tillamook Bay and the Nehalem River Estuary in Oregon, and Grays Harbor, and Willapa Bay in Washington.

Human Activities and their Impacts

Land Use. The rugged topography of the western Olympic Peninsula and the Oregon Coastal Range has limited the development of dense population centers. For instance, the Nehalem River and the Umpqua River basins consist of less than 1% urban land uses. Most basins in this region have long been exploited for timber production, and are still dominated by forestlands. In Washington State, roughly 90% of the coastal region is forested (Palmisano *et al.* 1993a). Approximately 92% of the Nehalem River basin is forested, with only 4% considered agricultural (Maser and Johnson 1999). Similarly, in the Umpqua River basin about 86% is forested land, 5% agriculture and 0.5% are considered urban lands—with about half the basin under Federal management (Carter and Resh 2005).

Tillamook County boasts about its dairy farming and cheese production—having a higher density of cows than people but even so, Tillamook County like many others in the region is dominated by forested lands (EPA 2006a). Roughly 90% of Tillamook County is forestland, held by Federal and state governments and private entities. In the Nehalem Basin, state and private landowners own more than 90% of the forestlands, and about 80% of the private land holdings are large timber companies (Maser and Johnson 1999).

Hydromodification Projects. Compared to other areas in the greater Northwest Region, the coastal region has fewer dams and several rivers remain free flowing (e.g., Clearwater River). The Umpqua River is fragmented by 64 dams, the fewest number of dams on any large river basin in Oregon (Carter and Resh 2005). According to Palmisano *et al.* (1993b) only about 30 miles of salmon habitat are permanently blocked by dams in the coastal streams of Washington.

In the past, temporary splash dams were constructed throughout the region to transport logs out of mountainous reaches. The general practice involved building a temporary dam in the creek adjacent to the area being logged, the pond was filled with logs and when the dam broke the floodwater would carry the logs to downstream reaches where they could be rafted and moved to market or downstream mills. Thousands of splash dams were constructed across the Northwest in the late 1800s and early 1900s. While the dams typically only temporarily blocked salmon habitat, in some cases they remained long enough to wipe out entire runs, the effects of the channel scouring and loss of channel complexity resulted in the long term loss of salmon habitat (NRC 1996a).

Mining. Oregon is ranked 35th nationally in total nonfuel mineral production value in 2004, while Washington was ranked 13th nationally in total nonfuel mineral production value 2004 and 17th in coal production (Palmisano *et al.* 1993a, NMA 2007). Metal mining for all metals (e.g., zinc, copper, lead, silver, and gold) peaked in Washington between 1940 and 1970 (Palmisano *et al.* 1993a). Today, construction sand and gravel, Portland cement and crushed stone are the predominant materials mined in both Washington and Oregon. Where sand and gravel is mined from riverbeds (gravel bars and floodplains) it may result in changes in channel elevations and patterns, instream sediment loads, and seriously alter instream habitat. In some cases, instream or floodplain mining has resulted in large scale river avulsions. The effect of mining in a stream or reach depends upon he rate of harvest and the natural rate of replenishment, as well as flood and precipitation conditions during or after the mining operations.

Commercial and Recreational Fishing. Most of the commercial landings in the region are groundfish, Dungeness crab, shrimp, and salmon. Many of the same species are sought by Tribal fisheries, and by charter, and recreational anglers. Nets and trolling are used in commercial and Tribal fisheries, whereas recreational anglers typically use hook and line, and may fish from boat, river bank, and docks. Entanglement of marine mammals in fishing gear is not uncommon and can lead to mortality or serious injury.

Marine Systems (Bioregions NA-S1, SP-IX, SP-XII, SP-XXI, NEP-I, NEP-II)

Hurricanes

The Gulf of Mexico, Caribbean Sea, and southern US Atlantic seaboard is prone to major tropical weather systems, including tropical storms and hurricanes. The impacts of these storms on sea turtles in the marine environment is not known, but storms can cause major impacts to sea turtle eggs on land, as nesting frequently overlaps with hurricane season, particularly Kemp's ridley sea turtles (NRC 1990a). Mortality can result both from drowning of individuals while still in the egg or emerging from the nest as well as causing major topographic alteration to beaches, preventing hatchling entry to marine waters. Kemp's ridley sea turtles are likely highly sensitive to hurricane impacts, as their only nesting locations are in a limited geographic area along southern Texas and northern Mexico (Milton *et al.* 1994). In 2010, Hurricane Alex made landfall in this area; surprisingly, few nests were lost (Jaime Pena, Gladys Porter Zoo, pers. comm.). Tropical storm Hermine arrived too late in 2010 to impact eggs or hatchlings at Rancho Nuevo (Donna Shaver, NPS, pers. comm.).

Habitat degradation

A number of factors may be directly or indirectly affecting listed species in the action area by degrading habitat; perhaps most significant among them is anthropogenic noise in the ocean. Natural sources of ambient noise include: wind, waves, surf noise, precipitation, thunder, and biological noise from marine mammals, fishes, and crustaceans. Anthropogenic sources of ambient noise include: transportation and shipping traffic, dredging, construction activities, geophysical surveys, and sonars. In general, it has been asserted that ocean background noise levels have doubled every decade for the last six decades in some areas, primarily due to shipping traffic (IWC 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 (Hatch *et al.* 2008).

Vessel noise could affect marine animals in the action area. Shipping and seismic noise generally dominates ambient noise at frequencies from 20 to 300 Hz (Richardson *et al.* 1995b, Andrew *et al.* 2002, Hildebrand 2009). 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 (NRC 1994, Richardson *et al.* 1995b, NRC 2003, Jasny *et al.* 2005, NRC 2005, McDonald *et al.* 2006, Hildebrand 2009). Over the past 50 years, the number of commercial vessels has tripled, carrying an estimated six times as much cargo (requiring larger, more powerful, and consequently louder vessels) (Hildebrand 2009). Seismic signals also contribute significantly to the low frequency ambient sound field (Hildebrand 2009). Baleen whales may be more sensitive to sound at those low frequencies than are toothed whales. Dunlop *et al.* (2010) found that humpback whales shifted from using vocal communication (which carries relatively large amounts of information) to surface-active communication (splashes; carry relatively little information) when low-frequency background noise increased due to increased sea state. Sonars and small vessels also contribute significantly to mid-frequency ranges (Hildebrand 2009).

The northeastern US hosts some of the busiest commercial shipping lanes in the world, including those leading into Boston, Providence, Newark, and New York (MARAD 2011). In addition to vessel traffic, marine construction activities occur in the Cape Cod area (liquefied natural gas terminal construction, pile driving, offshore wind farm construction, dredging, cable laying, drilling, and others) that contributes to local and regional background sound levels.

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 US (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 US 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 US, accounting for 6% of the total cargo entering the US, and increasing rapidly (growing 122% between 2003 and 2006) (DoT 2007b, a). Los Angeles is also the fifth largest cruise ship terminal in the US. A shipping lane runs along the US west coast south to southern California and additional shipping lanes extend westward from San Francisco and near Santa Barbara Island. While there are many major ports along the Pacific Coast, the states of Washington, Oregon, and California have proposed more stringent ballast water discharge standards than those proposed by the USCG's ballast water management program. Upon initial review, the USCG determined these standards were not practicable and to date they have not been implemented. As part of the USCG's ballast water management program, they will assess the performance of all technologies every three years in their practicability and periodic reviews to determine whether it is practicable to implement a more stringent ballast water standard.

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 proposed and listed species 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, Reyff 2003, Illingworth and Rodkin Inc. 2004).

Dredging operations also have the potential to emit sounds at levels that could disturb individuals of many taxa. 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). Three large, in-water construction projects are known to be underway or underdevelopment that have the potential to impact North Atlantic right whales. A 130-turbine wind farm is proposed or reviewed for construction off Long Island, New York and another in Massachusetts Bay. Both projects would involve pile driving that produces large amounts of sound in the frequency range used by North Atlantic right whales. As this sound would likely persist for extended periods, there is the potential for North Atlantic right whales to abandon local areas in favor of areas where they can better used their primary mechanism for gaining information about their environment. Although neither area co-occurs locally with high North Atlantic right whale use, individuals do forage in waters near the Massachusetts site and may migrate past the New York site to and/or from their southeastern/northeastern US breeding and foraging areas. In addition, a liquefied natural gas terminal is planned for construction off Gloucester, Massachusetts. This project would involve similar stressors, but located in prime North Atlantic right whale foraging habitat.

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

The state of marine and estuarine wetlands are a particular concern because of the reliance that listed anadromous fish have on them (Quinn 2005). Wetlands are the most diverse and productive systems in the marine environment, supporting 80% of Earth's fish and shellfish through feeding, nesting, shelter, high tide refuge, spawning grounds, and nursery habitat (Bertness 1999, RAE/ERF 1999, Airamé *et al.* 2003, Quinn 2005). Washington's wetlands serve as designated critical habitat for one-third of the state's listed species, including ESA-listed salmonids (Lane and Taylor 1997).

Ingestion of marine debris can have fatal consequences even for large whales as well as sea turtles. In 1989, a stranded sperm whale along the Mediterranean was found to have died from ingesting plastic that blocked its' digestive tract (Viale *et al.* 1992). A sperm whale examined in Iceland had a lethal disease thought to have been caused by the complete obstruction of the gut with plastic marine debris (Lambertsen 1990). The stomach contents of two sperm whales that stranded separately in California included extensive amounts of discarded fishing netting (NMFS 2009). A fifth individual from the Pacific was found to contain nylon netting in its stomach when it washed ashore in 2004 (NMFS 2009). Further incidents may occur but remain undocumented when carcasses do not strand.

For sea turtles, marine debris is a problem due primarily to individuals ingesting debris and blocking the digestive tract, causing death or serious injury (Lutcavage *et al.* 1997a, Laist *et al.* 1999). Gulko and Eckert (2003) estimated that between one-third and one-half of all sea turtles ingest plastic at some point in their lives; this figure is supported by data from Lazar and Gracan (Lazar and Gračan 2010), who found 35% of loggerheads had plastic in their gut. A Brazilian study found that 60% of stranded green sea turtles had ingested marine debris (primarily plastic and oil; (Bugoni *et al.* 2001)). Loggerhead sea turtles had a lesser frequency of marine debris ingestion. Plastic is possibly ingested out of curiosity or due to confusion with prey items; for example, plastic bags can resemble jellyfish (Milton and Lutz 2003). Marine debris consumption has been shown to depress growth rates in post-hatchling loggerhead sea turtles, elongating the time required to reach sexual maturity and increasing predation risk (McCauley and Bjorndal 1999). Studies of shore cleanups have found that marine debris washing up along the northern Gulf of Mexico shoreline amounts to about 100 kg/km (ACC 2010, LADEQ 2010, MASGC 2010, TGLO 2010). Sea turtles can also become entangled and die in marine debris, such as discarded nets and monofilament line (O'Hara *et al.* 1988, NRC 1990a, Lutcavage *et al.* 1997a, Laist *et al.* 1999). This fundamentally reduces the reproductive potential of affected populations, many of which are already declining (such as loggerhead and leatherback sea turtle populations in the action area).

Invasive species have had major impacts on aquatic, estuarine, and marine ecosystems. As information presented here would be largely duplicative of that presented in the *Effects analysis*, we detail the impacts of invasive species to listed resources in the *Effects analysis*.

Oil and Gas Development

The Arctic Ocean off Alaska and Gulf of Mexico are the epicenters for marine oil and gas development and extraction within the action area. The Arctic's major sources of industrial underwater noise appear to be offshore oil, gas or mineral exploration and exploitation. These activities increase vessel traffic, produce loud sounds for seismic profiling, place structures in areas used by whales, and introduce noises from drilling and production into the environment (NMFS 1999, 2006f). Malme *et al.* (1985) exposed feeding humpback whales in southeastern Alaska to noise from a single air gun or to playback of recorded sounds of oil drilling, production platforms and aircraft. Whales showed no overall pattern of avoidance during 13 experiments, each of which included between 10 and 40 different animals. Whales startled as soon as the airgun was turned on in three experiments. These startle responses, which occurred at received sound levels between 150 to 169 dB (re 1 mPa), were thought to be caused more by the novelty of the air gun sound than by its intensity. There are presently two offshore production facilities within state waters in the Beaufort Sea: Northstar and Endicott. Multiple marine geophysical (seismic) projects are planned for the Beaufort and Chukchi Seas.

The northern Gulf of Mexico is the location of massive industrial activity associated with oil and gas extraction and processing. Over 4,000 oil and gas structures are located outside of state waters in the northern Gulf of Mexico; 90% of these occur off Louisiana and Texas (USN 2009). This is both detrimental and beneficial for sea turtles. These structures appreciably increase the amount of hard substrate in the marine environment, providing shelter and foraging opportunities for species like loggerhead sea turtles (Parker *et al.* 1983, Stanley and Wilson 2003). However, the Bureau of Ocean Energy Management requires that structures must be removed within one year of lease termination. Many of these structures are removed by explosively severing the underwater supportive elements, which produces a shock wave that kills, injures, or disrupts marine life in the blast radius (Gitschlag *et al.* 1997). For sea turtles, this means death or serious injury for individuals within a few hundred meters of the structure and overt behavioral (potentially physiological) impacts for individuals further out (Duronslet *et al.* 1986, Klima *et al.* 1988). Although observers and procedures are in place to mitigate impacts to sea turtles (i.e., not blasting when sea turtles are present), not all sea turtles are observed all the time and low-level sea turtle injury and mortality still occurs (Gitschlag and Herczeg 1994, Gitschlag *et al.* 1997); two loggerheads were killed in August 2010 (G. Gitschlag, NOAA, pers. comm.). Current annual authorized takes due to Bureau of Ocean Energy Management OCS oil and gas exploration, development, production, and abandonment activities are 30 sea turtles, including no more than one each of Kemp's ridley, green, or hawksbill turtles and no more than ten loggerhead turtles (NMFS 1988). These levels were far surpassed by the *Deepwater Horizon* incident (see oil spills and releases below). Overall, these activities provide both positive and negative effects at the individual level and have no clear impact at the population and species levels.

Oil pollution has been a significant concern in the Gulf of Mexico for several decades due to the large amount of extraction and refining activity in the region. Routine discharges into the northern Gulf of Mexico (not including oil spills) include roughly 88,200 barrels of petroleum per year from municipal and industrial wastewater treatment plants and roughly 19,250 barrels from produced water discharged overboard during oil and gas operations (MMS 2007b, USN 2008). These sources amount to over 100,000 barrels of petroleum discharged into the northern Gulf of Mexico annually. Although this is only 10% of the amount discharged in a major oil spill, such as the Exxon *Valdez* spill (roughly 1 million barrels), this represents a significant, continual, and “unseen” threat to Gulf of Mexico wildlife and habitats. Generally, accidental oil spills may amount to less than 24,000 barrels of oil discharged annually in the northern Gulf of Mexico, making non-spilled oil normally one of the leading sources of oil discharge into the Gulf of Mexico, although incidents such as the 2010 *Deepwater Horizon* incident are exceptional (MMS 2007a). The other major source from year to year is oil naturally seeping into the northern Gulf of Mexico. Although exact figures are unknown, natural seapage is estimated at between 120,000 and 980,000 barrels of oil annually (MacDonald *et al.* 1993, MMS 2007b).

Although non-spilled oil is the primary contributor to oil introduced into the Gulf of Mexico, concern over accidental oil spills is well-founded. Over five million barrels of oil and one million barrels of refined petroleum products are transported in the northern Gulf of Mexico daily (MMS 2007b); worldwide, it is estimated that 900,000 barrels of oil are released into the environment as a result of oil and gas activities (Epstein and (Eds.). 2002). Even if a small fraction of the annual oil and gas extraction is released into the marine environment, major, concentrated releases can result in significant environmental impacts. Due to the density of oil extraction, transport, and refining facilities in the Houston/Galveston and Mississippi Delta areas (and the extensive activities taking place at these facilities), these locations have the greatest probability of experiencing oil spills. Oil released into the marine environment contains aromatic organic chemicals known to be toxic to a variety of marine life; these chemicals tend to dissolve into the air to a greater or lesser extent, depending upon oil type and composition

(Yender *et al.* 2002). Solubility of toxic components is generally low, but does vary and can be relatively high (0.5-167 parts per billion; (Yender *et al.* 2002)). Use of dispersants can increase oil dispersion, raising the levels of toxic constituents in the water column, but speeding chemical degradation overall (Yender *et al.* 2002). The remaining oil becomes tar, which forms floating balls that can be transported thousands of kilometers into the North Atlantic. The most toxic chemicals associated with oil can enter marine food chains and bioaccumulate in invertebrates such as crabs and shrimp to a small degree (prey of some sea turtles (Marsh *et al.* 1992, Law and Hellou 1999)), but generally do not bioaccumulate or biomagnify in finfish (Varanasi *et al.* 1989, Meador *et al.* 1995, Baussant *et al.* 2001, Yender *et al.* 2002). The loss of invertebrate communities due to oiling or oil toxicity would also decrease prey availability for hawksbill, Kemp's ridley, and loggerhead sea turtles (NOAA 2003). Furthermore, Kemp's ridley and loggerhead sea turtles, which commonly forage on crustaceans and mollusks, may ingest large amounts of oil due oil adhering to the shells of these prey and the tendency for these organisms to bioaccumulate toxins found in oil (NOAA 2003). It is suspected that oil adversely impacted the symbiotic bacteria in the gut of herbivorous marine iguanas when the Galapagos Islands experienced an oil spill, contributing to a >60% decline in local populations the following year. The potential exists for green sea turtles to experience similar impacts, as they also harbor symbiotic bacteria to aid in their digestion of plant material (NOAA 2003). Seagrass beds may be particularly susceptible to oiling as oil contacts grass blades and sticks to them, hampering photosynthesis and gas exchange (Wolfe *et al.* 1988). If spill cleanup is attempted, mechanical damage to seagrass can result in further injury and long-term scarring. Loss of seagrass due to oiling would be important to green sea turtles, as this is a significant component of their diets (NOAA 2003). Sea turtles are known to ingest and attempt to ingest tar balls, which can block their digestive systems, impairing foraging or digestion and potentially causing death (NOAA 2003). Dispersants reduce the formation of tar balls. Although the effects of dispersant chemicals on sea turtles is unknown, testing on other organisms have found currently used dispersants to be less toxic than those used in the past (NOAA 2003). It is possible that dispersants can interfere with surfactants in the lungs (surfactants prevent the small spaces in the lungs from adhering together due to surface tension, facilitating large surface areas for gas exchange), as well as interfere with digestion, excretion, and salt gland function (NOAA 2003). Oil exposure can also cause acute damage upon direct exposure to oil, including skin, eye, and respiratory irritation, reduced respiration, burns to mucous membranes such as the mouth and eyes, diarrhea, gastrointestinal ulcers and bleeding, poor digestion, anemia, reduced immune response, damage to kidneys or liver, cessation of salt gland function, reproductive failure, and death (Vargo *et al.* 1986c, b, a, NOAA 2003, 2010b). Nearshore spills or large offshore spills can oil beaches on which sea turtles lay their eggs, causing birth defects or mortality in the nests (NOAA 2003, 2010b).

Several oil spills have impacted the northern Gulf of Mexico over the past few years, largely due to hurricanes. The impacts of Hurricane Ivan in 2004 on the Gulf Coast included pipeline damage causing 16,000 barrels of oil to be released and roughly 4,500 barrels of petroleum products from other sources (USN 2008, BOEMRE 2010). The next year, Hurricane Katrina caused widespread damage to onshore oil storage facilities, releasing 191,000 barrels of oil (LHR 2010). Another 4,530 barrels of oil were released from 70 other smaller spills associated with hurricane damage. Shortly thereafter, Hurricane Rita damaged offshore facilities resulting in 8,429 barrels of oil to be released (USN 2008).

Major oil spills have impacted the Gulf of Mexico for decades (NMFS 2010). Until 2010, the largest oil spill in North America occurred in the Bay of Campeche (1979), when a well “blew out”, allowing oil to flow into the marine environment for nine months, releasing 2.8-7.5 million barrels of oil. Oil from this release eventually reached the Texas coast, including the Kemp’s ridley sea turtle nesting beach at Rancho Nuevo, from where 9,000 hatchlings were airlifted and released offshore (NOAA 2003). Over 7,600 m^3 of oiled sand was eventually removed from Texas beaches and 200 gallons of oil were removed from the area around Rancho Nuevo (NOAA 2003). Eight dead and five live sea turtles were recovered during the oil spill event; although cause of deaths were not determined, oiling was suspected to play a part (NOAA 2003). Also in 1979, the oil tanker *Burmah Agate* collided with another vessel near Galveston, Texas, causing an oil spill and fire that ultimately released 65,000 barrels of oil into estuaries, beachfronts, and marshland along the northern and central Texas coastline (NMFS 2010). Clean-up of these areas was not attempted due to the environmental damage such efforts would have caused. Another 195,000 barrels of oil are estimated to have been burned in a multi-month-long fire aboard the *Burmah Agate* (NMFS 2010). The tanker *Alvenus* grounded in 1984 near Cameron, Louisiana, spilling 65,500 barrels of oil which spread west along the shoreline to Galveston (NMFS 2010). One oiled sea turtle was recovered and released (NOAA 2003). In 1990, the oil tanker *Megaborg* experienced an accident near Galveston during the lightering process and released 127,500 barrels of oil, most of which burned off in the ensuing fire (NMFS 2010).

On April 20, 2010, a fire and explosion occurred aboard the semisubmersible drilling platform *Deepwater Horizon* roughly 80 km southeast of the Mississippi Delta (NOAA 2010a). The platform had 17,500 barrels of fuel aboard, which likely burned, escaped, or sank with the platform (NOAA 2010a). However, once the platform sank, the riser pipe connecting the platform to the wellhead on the seafloor broke in multiple locations, initiating an uncontrolled release of oil from the exploratory well. Over the next three months, oil was released into the Gulf of Mexico, resulting in oiled regions of Texas, Louisiana, Mississippi, Alabama, and Florida and widespread oil slicks throughout the northern Gulf of Mexico that closed more than one-third of the Gulf of Mexico Exclusive Economic Zone to fishing due to contamination concerns. Apart from the widespread surface slick, massive undersea oil plumes formed, possibly through the widespread use of dispersants, and reports of tarballs washing ashore throughout the region were common. Although estimates vary, NOAA has estimated that 4.9 million barrels of oil were released (Lubchenco *et al.* 2010). A total of 720 sea turtles have been verified in the spill zone of which 172 were verified as having been exposed to oil (NOAA 2010c). However, specific causes of injury or death have not yet been established for many individuals as investigations into the role of oil in these animals’ health status continue.

Perhaps the most famous shipwreck of all time occurred in the Gulf of Alaska when, in 1989, the *Exxon Valdez* released at least 11 million gallons of Alaskan crude oil into one of the largest and most productive estuaries in North America. The spill was the worst in US 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 US shores at refining and transfer facilities and extraction sites.

Sea turtles may also be harassed by the high level of helicopter activity over Gulf of Mexico waters. It is estimated that between roughly 900,000 and 1.5 million helicopter take-offs and landings are undertaken in association with oil and gas activities in the Gulf of Mexico annually (NRC 1990a, USN 2008). This likely includes numerous overflights of sea turtles, an activity which has been observed to startle and at least temporarily displace sea turtles (USN 2009).

Pollution

Chemical pollution of the freshwater, estuarine, and marine environment is a pervasive problem throughout the US, although the significance of specific pollutants varies between regions or watersheds. The Gulf of Mexico is a sink for massive levels of pollution from a variety of marine and terrestrial sources, which ultimately can interfere with ecosystem health and particularly that of sea turtles (see *Status of listed resources* section). Sources include the petrochemical industry in and along the Gulf of Mexico, wastewater treatment plants, septic systems, industrial facilities, agriculture, animal feeding operations, and improper refuse disposal. The Mississippi River drains 80% of United States cropland (including the fertilizers, pesticides, herbicides, and other contaminants that are applied to it) and discharges into the Gulf of Mexico near the action area (MMS 1998). Agricultural discharges, as well as discharges from large urban centers (ex.: Houston and New Orleans) contribute contaminants as well as coliform bacteria to Gulf of Mexico habitats (Garbarino *et al.* 1995). These contaminants can be carried long distances from terrestrial or nearshore sources and ultimately accumulate in offshore pelagic environments (USCOP 2004). The ultimate impacts of this pollution are poorly understood.

Significant attention has been paid to nutrient enrichment of Gulf of Mexico waters, which leads to algal blooms (including harmful algal blooms), oxygen depletion, loss of seagrass and coral reef habitat, and the formation of a hypoxic "dead zone" (USCOP 2004). This hypoxic event occurs annually from as early as February to as late as October, spanning roughly 12,700 km^2 (although in 2005 the "dead zone" grew to a record size of 22,000 km^2) from the Mississippi River Delta to Galveston, Texas (MMS 1998, Rabalais *et al.* 2002, LUMCON 2005, USGS 2010). Although sea turtles do not extract oxygen from sea water, numerous staple prey items of sea turtles, such as fish, shrimp, and crabs, do and are killed by the hypoxic conditions (Craig *et al.* 2001). More generally, the "dead zone" decreases biodiversity, alters marine food webs, and destroys habitat (Craig *et al.* 2001, Rabalais *et al.* 2002). High nitrogen loads entering the Gulf of Mexico from the Mississippi River is the likely culprit; nitrogen concentrations entering the Gulf of Mexico have increased three fold over the past 60 years (Rabalais *et al.* 2002). Through these indirect effects, sea turtles are unable to utilize this region during this time for foraging and can only utilize it to a limited extent when the "dead zone" does not occur while the underlying food web recovers.

Due to rapid human population growth and urban development in the Puget Sound region (Grant and Ross. 2002), significant pollutant loads have led to generally poor water quality in the region, including Puget Sound itself (Long *et al.* 2001, Grant and Ross. 2002, NMFS 2008e). Some of these pollutants have apparently bioaccumulated in southern resident killer whales, with persistent organic pollutants now being found in high levels in killer whales along British Columbia and Washington State (Koski 2007b) and considered to pose the greatest pollution threat to southern resident killer whales (Ross *et al.* 2000c, CBD 2001a, Krahn *et al.* 2002b, Koski 2007b, NMFS 2008e). These pollutants have the potential to cause immune, endocrine, and reproductive system effects in killer whales; concern has been expressed by researchers that these pollutants may be a factor in the decline of southern resident killer whales (Calambokidis *et al.* 1984, Hayteas and Duffield. 2000, Ross *et al.* 2000b, Ross *et al.* 2000c, Baird 2001b, Ylitalo *et al.* 2001, de Wit 2002, Krahn *et al.* 2002b, Darnerud 2003, Hall *et al.* 2003, Krahn *et al.* 2004b, Krahn *et al.* 2004c, Waring *et al.* 2004a, Krahn *et al.* 2007, NMFS 2008e). Exposure is believed to be primarily through diet (Hickie *et al.* 2001); salmon preyed upon by southern resident killer whales may also be at risk. In addition, several hundred new chemicals enter the global marine environment annually; many of these chemicals have unknown effects to any lifeform (Grant and Ross. 2002, NMFS 2008e). Encouragingly, many persistent organic pollutants have been banned in the US and, over the past few decades, regulatory actions, Superfund clean-up, improved waste handling, and ongoing cleanup efforts have led to improvements in regional water quality (NMFS 2008e). This has led to decreasing levels of many organochlorine residues in the environment (Gray and Tuominen 2001, Mearns 2001, Grant and Ross. 2002, EVS Environmental Consultants 2003), although it may take up to 60 years for some chemicals to fall below levels known to cause health effects in marine mammals (Hickie *et al.* 2001).

Another significant form of pollution is from the petrochemical industry utilizing the Puget Sound region for transport and refining. Puget Sound is one of the leading petroleum refining centers in the US with about 333 million barrels of crude oil and refined petroleum products transported through it annually (Puget Sound Action Team 2005). Inbound oil tankers carry crude oil to five major refineries in the sound, while outbound tankers move refined oil products to destinations along the US west coast (Neel *et al.* 1997). In 2005, a total of 716 oil tankers passed through Washington's waters bound for ports in Puget Sound, Canada, and along the Columbia River (WSDE 2006). In general, the Strait of Juan de Fuca and areas near Washington's major refineries (located in Anacortes, Ferndale, Blaine, and Tacoma) are considered the locations most at risk of major spills in the action area (Neel *et al.* 1997, NMFS 2008e). Since the 1960s, there have been at least nine major oil spills of at least 2,222 barrels; the largest was over 50,000 barrels (Neel *et al.* 1997, Puget Sound Water Quality Action Team 2002).

In addition to the substantial volume of shipping traffic and the prevalence of petroleum refining centers in the action area, of equal or perhaps greater concern are the chronic small-scale discharges of oil, which together can be substantial (Clark 1997). These discharges originate from numerous sources, including the dumping of tank washings and ballast water by tankers, the release of bilge and fuel oil from general shipping, and the disposal of municipal and industrial waters. Chronic oil pollution kills large numbers of seabirds (e.g., (Wiese and Robertson 2004)); however, its impact on southern resident killer whales and other marine mammals is poorly documented (NMFS 2008e). In addition, the long-term effects of repeated ingestion of sub-lethal quantities of petroleum hydrocarbons on marine mammals are not well understood. As a result, the magnitude of the risks posed by oil discharges in the proposed action area is difficult to precisely quantify or estimate.

Commercial Fisheries

Commercial fisheries are a major factor in the decline of several listed species in this Opinion.

Directed harvesting of whales as been the greatest single reason several whale species are listed as threatened or endangered. However, harvests of listed whales have been largely eliminated and the species dealt with in this Opinion species appear to be generally recovering. The significance of whale reductions compared to historical levels is provided under *Status of Listed Resources*. Salmonids have also experienced significant fishing pressure, which is discussed below. Sea turtle species also continue to suffer from commercial harvesting in their nesting areas. While there is an international moratorium on harvesting whales, Japan harvests up to 101 sei whales annually (IWC 2008).

Although protected under the ESA, listed salmonids are killed in commercial harvests along the US west coast. As salmonid DPSs/ESUs are readily differentiated genetically, but not morphologically, the extent of impact that harvesting has is unknown, although documentation of species-level mortality is. Chinook and Coho salmon have historically been the primary salmonids caught along the Oregonian coast, although in 2002, Chinook salmon comprised only 2.4% (2,247 metric tons) of the all landings and Coho salmon was only 0.5% (526 metric tons) due to crashes in both species (State 2004). In Washington, salmon accounted for ~15% of commercial fishery landings in 2007. This included 1,377 metric tons of Chinook, 5,352 metric tons of chum, 1,139 metric tons of Coho, 36 metric tons of sockeye, and 147 metric tons of steelhead (WDFW 2007). Sockeye and steelhead are not subject to directed fisheries in the area (PFMC 2005). Along both Oregon and Washington, harvests generally occur over the continental shelf within 65 km of shore and generally operate from March-November (NMFS-NWR 2003). Across the entire US west coast, chum and Chinook salmon dominate commercial harvesting, with Coho composing a smaller component (Figure 24).

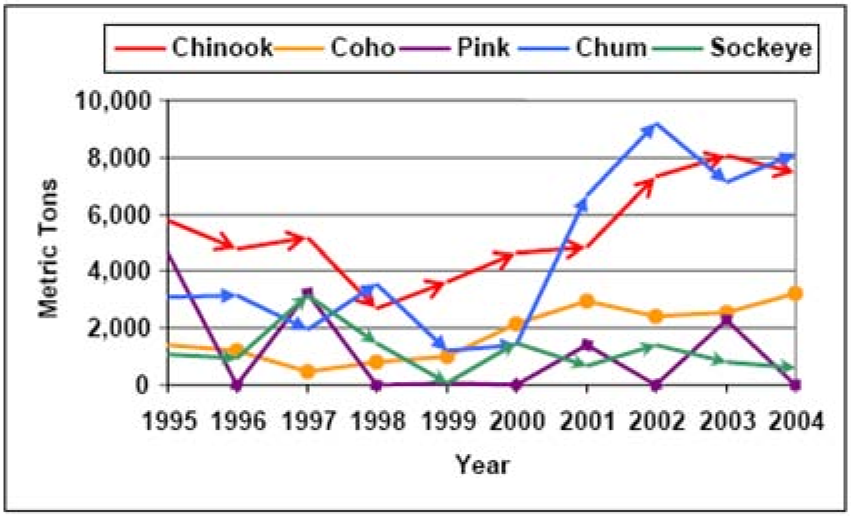

Figure 24. Annual landings of Chinook, Coho, pink, chum, and sockeye salmon for California, Oregon, and Washington State between 1995 and 2004 (from USN (2006)).

Apart from direct and bycatch mortality, there are additional consequences of fishing. Trawls can disturb the ocean floor, causing physical damage to exposed structures, increase turbidity, alter sediments, and remove prey of listed species. Fishing gear can also be lost, but subsequently entangle or trap fish and marine mammals years later. Recreational fishing can also reduce populations by focusing on certain species or locations.

The Gulf of Alaska supports one of the most productive ecosystems in the US and is host to a very large fishing industry. The Alaskan groundfish fishery has received considerable attention for its exploitation of some fish species that are also prey to Steller sea lions. Significant concern exists that the level of fishing results in significant prey reductions in the region, limiting prey availability for Steller sea lions. In 2005, 2006, and 2007, 38,000-48,000 metric tons of pollack, 20,000-28,000 metric tons of Pacific cod, 25,000 metric tons of arrowtooth flounder, 8,000-8,500 metric tons of rockfish, and ~5,500 metric tons of skates were allowed to be harvested annually (NMFS 2006a).

The waters surrounding Washington State host extensive commercial and recreational fishing; in 2007, 180 million pounds of fish and invertebrates were harvested from state waters; of this, 15% was salmon (Commander U.S. Pacific Fleet 2010). Although closely regulated, there is the potential for overfishing, especially considering the selective nature of southern resident killer whale diets (Fraser River populations of Chinook salmon). Lost gill nets, purse seines, and longlines may foul and disrupt bottom habitats as well as create the potential for "ghost fishing," where marine animals are continually trapped and die in derelict gear. Recent reports of entanglement are unknown, but Sheffer and Slipp (1948a) documented several deaths of animals caught in gillnets between 1929 and 1943. 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 (Scheffer and Slipp. 1948a, Pike and Macaskie. 1969, Haley 1970, Olesiuk *et al.* 1990d, Baird 2001b). This practice has largely abated in the past few decades and is unlikely to occur today (Young *et al.* 1993, Carretta *et al.* 2001).

Three of the biggest threats to sea turtles result from harvest for commercial and subsistence use. These include egg harvest, the harvest of females on nesting beaches, and directed hunting of sea turtles in foraging areas. These factors have led to the precipitous declines in worldwide sea turtle populations. In the Atlantic, green sea turtles are captured and killed in turtle fisheries in Colombia, Grenada, the Lesser Antilles, Nicaragua, St. Vincent and the Grenadines; the turtle fishery along the Caribbean coast of Nicaragua, by itself, has captured more than 11,000 green sea turtles annually over the past decade (Lagueux 1998, Bräutigam and Eckert 2006a). While these threats have been largely eliminated in Florida due to successful conservation measures, the hunting of juvenile and adult turtles continues both legally and illegally in many foraging areas where green sea turtles originating from Florida are known to occur (Fleming 2001, Chacon 2002). The killing of nesting hawksbill females continues to threaten the stability of hawksbill subpopulations in many areas. The centuries-old historic trade in tortoise shell greatly impacted hawksbill populations in the Insular Caribbean. Increases in nesting hawksbills in the region coincide with the decline of international trade in hawksbill shell (Milliken and Tokunaga 1987), and in particular with the 90% reduction in the annual take of large hawksbills from Cuban waters (Carrillo *et al.* 1999).

Entrapment and Entanglement in Fishing Gear

Fisheries interactions are a significant problem for several marine mammal species and particularly so for humpback whales. 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 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 US 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. A photography study of humpback whales in southeastern Alaska in 2003 and 2004 found at least 53% of individuals showed some kind of scarring from fishing gear entanglement (Neilson *et al.* 2005).

Fisheries interactions are a particularly problem for North Atlantic right whales (Figure 25). 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. Along the Atlantic coast of the US and the Maritime Provinces of Canada, there were 46 confirmed reports of North Atlantic right whales entangled in fishing gear between 1990 and 2007 (Cole *et al.* 2005a, Nelson *et al.* 2007a, Waring *et al.* 2009). Of the 39 reports that the NMFS could confirm, North Atlantic right whales were injured in five of the entanglements and killed in four entanglements. Three of the 24 entangled whales between 2004 and 2008 died and one other resulted in serious injury (Glass *et al.* 2009). Recent efforts to disentangle right whales have met with success (Anonmyous. 2009b). However, over 60% of the North Atlantic right whale population show some evidence of entanglement (Hamilton *et al.* 1998).

Figure 25. A North Atlantic right whale entangled in fisheries gear off Florida, with Georgia Department of Natural Resources and Coastwise Consulting staff attempting to cut rope off (Credit: EcoHealth Alliance and Georgia Department of Natural Resources, ESA permit number 932-1905).

Recent reports of entanglement are unknown, but Sheffer and Slipp (1948b) 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 (Scheffer and Slipp. 1948b, Pike and Macaskie. 1969, Haley 1970, Olesiuk *et al.* 1990c, Baird 2001c). This practice has largely abated in the past few decades and unlikely to continue today (Young *et al.* 1993, Carretta *et al.* 2001).

Steller sea lions routinely become entangled in fishing gear, but are most prone to entanglement in packing and rubber bands around their necks (Raum-Suryan *et al.* 2009b)(Figure 26). As an individual grows, these items eventually cut into tissue, leading to gruesome injuries and, frequently, death. Several dozen individuals may become entangled and drown in commercial fishing gear annually (Atkinson *et al.* 2008, NMFS 2008g). Marine debris is also concerning for the health of Steller sea lion populations. It is estimated that 0.26% of Steller sea lions have marine debris around their necks or are hooked by fishing gear (0.07%); this equates to 100-200 individuals annually (FOC 2008, Raum-Suryan *et al.* 2009a). Individuals also commonly ingest flashers and hooks used in commercial fisheries (Raum-Suryan *et al.* 2009b). These can cause several internal injuries potentially leading to death.

Figure 26. 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).

Fishery interaction remains a major factor in sea turtle recovery and, frequently, the lack there of. NMFS (2002) estimated that 62,000 loggerhead sea turtles have been killed as a result of incidental capture and drowning in shrimp trawl gear. Although turtle excluder devices and other bycatch reduction devices have significantly reduced the level of bycatch to sea turtles and other marine species in US waters, mortality still occurs in Gulf of Mexico waters.

In addition to commercial bycatch, recreational hook-and-line interaction also occurs. Cannon and Flanagan (1996) reported that from 1993 to 1995, at least 170 Kemp's ridley sea turtles were hooked or tangled by recreational hook-and-line gear in the northern Gulf of Mexico. Of these, 18 were dead stranded turtles, 51 were rehabilitated turtles, five died during rehabilitation, and 96 were reported as released by fishermen.

Naval Activities

Naval activity, notable 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 and sea turtles. The action area overlaps several naval training ranges or facilities listed below. Listed whales travel widely in the North Atlantic and could be exposed to naval activities in several ranges.

- Marianas Island Range Complex, where humpback whales breed and possibly give birth,
- The Southern California Range Complex,
- The Northwest Training Range Complex, where humpback whales and Steller sea lions forage,
- 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 and Hawaiian monk seals regularly breed and give birth.

- Northeast Training Range Complex, where North Atlantic right and humpback whales forage,
- The Virginia Capes, Cherry Point, and Jacksonville-Charleston Operating Areas, which are situated consecutively along the migratory corridor for North Atlantic right whales (the later operating area overlaps with North Atlantic right whale critical habitat, where calving and breeding take place), and
- The Key West, Gulf of Mexico, Bermuda, and Puerto Rican Complexes have the potential to overlap the distribution of North Atlantic right and humpback whales, but presence within these areas is rare or undocumented.

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.

Although naval vessels represent a small fraction of the total sound level and are designed to operate quietly, these ships are large and equipped with high-output sonar equipment such as ANISQS-53C tactical sonar, which produces signals at source levels of 235 dB re 1 µPa$_{rms}$ at 1 m. The signals emitted from these devices have the potential to affect marine mammals in the action area; however, empirical data are limited. 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. However, 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 US 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, 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, 2006b). Underwater detonations are sometimes performed at this site and there was an occasion when J pod was less than 1.5 km away when a blast occurred, which caused the whales to suddenly change their direction of travel (NMFS 2006b). In another incident, 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.

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 (Petras 2003, NMFS 2008e). However, these signals can also cause strong avoidance responses in cetaceans (Olesiuk *et al.* 2002). Morton and Symonds (2002) describe one AHD model that broadcasts a 10 kHz signal at 194 dB re 1 μPa at 1 m. A large majority of these occur in Arctic waters during exploration for petroleum products, although other detectable above ambient levels in open water for up to 50 km. Activation of AHDs at an aquaculture farm near northeastern Vancouver Island corresponded with drastic declines in the presence and use of nearby passages and inlets by both resident and transient killer whales (Morton and Symonds 2002). The only AHD still in use in Washington State operates at the Ballard locks in Seattle, where NMFS uses it to deter sea lions (NMFS 2008e).

Vessel Approaches – Commercial and Private Marine Mammal Watching

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

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

It is difficult to precisely quantify or estimate the magnitude of the risks posed to marine mammals in general and southern resident killer whales specifically (who possibly have the greatest exposure to whale watching activities of any listed marine mammal) by whale watching and recreational vessels (NMFS 2008e). Commercial whale watching in Washington State has increased dramatically from small scale operations during the late 1970s to early 1980s, to 13 vessels by 1988, and a total of 76 vessels (and over 500,000 people) in 2006 (Osborne 1991, Koski 2006a, Koski 2007b, NMFS 2008e). Most companies belong to the Whale Watch Operators Association Northwest, which has established whale viewing guidelines for commercial operators (WWOANW 2007). Currently, over 50% of vessels involved with whale watching are commercially owned, with the San Juan Islands and adjacent area also attracting large numbers of private boaters for recreational activities such as opportunistic viewing of killer whales (Koski 2007b, NMFS 2008e). In addition, private floatplanes, helicopters, and small aircraft regularly take advantage of whale watching opportunities (MMMP 2002b). Weather conditions in the Pacific Ocean in winter limit whale watching during winter months and activity is greatest during summer (NMFS 2008e). From May to September 2005, an average of over 19 boats (up to 94) surrounded southern resident killer whales on a daily basis (Koski 2006a). In Washington State, southern resident killer whales are the primary target species, particularly in Haro Strait (Hoyt 2001, 2002, NMFS 2008e).

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

Live-Captures for Aquaria

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

Ship-strike is a significant concern for the recovery of listed baleen whales and sea turtles to a lesser degree. 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 2003). Along the Pacific US 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 2003).

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

Ship-strike is a significant concern for the recovery of North Atlantic right whales in the region. We believe the vast majority of ship-strike mortalities go unnoticed, and that actual mortality is higher than currently documented; Kraus *et al.* (2005) estimated that 17% of ship strikes are actually detected. The magnitude of the risks commercial ship traffic pose to large whales in the proposed action areas has been difficult to quantify or estimate. We struggle to estimate the number of whales that are killed or seriously injured in ship strikes within the US EEZ and have virtually no information on interactions between ships and commercial vessels outside of US waters. With the information available, we know those interactions occur but we cannot estimate their significance to whale species.

Ship strikes are the largest single contributor to North Atlantic right whale deaths, accounting for approximately 35% of all known mortalities, even though right whales should be able to hear the sound produced by vessels (Richardson *et al.* 1995a, Ketten 1998, Knowlton and Kraus 2001, Laist *et al.* 2001). Some information suggests right whales respond only within very close proximity to ships (Nowacek *et al.* 2004). Injury is generally caused by the rotating propeller blades, but blunt injury from direct impact with the hull also occurs. There have been 18 reports of North Atlantic right whales being struck by vessels between 1999 and 2005 (Cole *et al.* 2005b, Nelson *et al.* 2007b). Of the 17 reports that NMFS could confirm, right whales were injured in two of the ship strikes and killed in nine. Recent records show that from 2004-2008, there were 17 confirmed reports of North Atlantic right whales being struck with eight whales dying of their wounds and two additional right whales sustaining serious injuries (Glass *et al.* 2009). Deaths of females are especially deleterious to the ability of the North Atlantic right whale population to recover. For instance, in 2005, mortalities included six adult females, three of which were carrying near-term fetuses and four of which were just starting to bear calves, thereby representing a lost reproductive potential of as many as 21 individuals over the short term (Kraus *et al.* 2005). Voluntary recommendations for slower vessel speeds in the Bay of Fundy appear to be largely ignored (Vanderlaan *et al.* 2008a). Rules for seasonal (varies by location, but January through July) slowing of vessel traffic to 10 knots or changing shipping lanes by less than one nautical mile to avoid the greatest concentrations of right whales are predicted to be capable of reducing ship strike mortality by 62% in the Bay of Fundy region; the same rule applies from November through April from Brunswick, Georgia to Jacksonville, Florida, where North Atlantic right whales go for calving and breeding. Speed rules also apply to medium and large ports along the eastern seaboard during this time frame when right whales migrate to and from northern feeding and southern breeding areas. Nearly a dozen shipping lanes transect through coastal waters of the southeastern US from the North-South Carolina to Cape Canaveral, Florida. The vast majority of ship strike mortalities are never identified, and that actual mortality is higher than currently documented.

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 (Visser 1999, Visser and Fertl. 2000, Baird 2001c, Carretta *et al.* 2001, Carretta *et al.* 2004).

Sea turtle ship strikes are a poorly-studied threat to sea turtles, but has the potential to be highly-significant (Work *et al.* 2010). All sea turtles must surface to breath and several species are known to bask at the surface for long periods, including loggerhead sea turtles. Although sea turtles can move rapidly, sea turtles apparently are not well able to move out of the way of vessels moving at more than 4 km/hr; most vessels move far faster than this in open water (Hazel and Gyuris 2006, Hazel *et al.* 2007, Work *et al.* 2010). This, combined with the massive level of vessel traffic in the Gulf of Mexico, has the potential to result in frequent injury and mortality to sea turtles in the region (MMS 2007b). Hazel *et al.* (2007) suggested that green sea turtles may use auditory cues to react to approaching vessels rather than visual cues, making them more susceptible to strike as vessel speed increases. Each state along the Gulf of Mexico has several hundred thousand recreational vessels registered, including Florida with nearly one million-the highest number of registered boats in the United States-and Texas with over 600,000- ranked sixth nationally (USCG 2003, 2005, NMMA 2007). Commercial vessel operations are also extensive. Vessels servicing the offshore oil and gas industry are estimated to make 115,675-147,175 trips annually, apart from commercial vessels travelling to and from some of the largest ports in the United States (such as New Orleans and Houston)(MMS 2007a, USN 2008). Extensive shrimping and other fishery effort is also expended in the area. Overall, ship strike is likely highly underestimated as a source of injury or mortality to sea turtles in the action area.

Scientific Research and Permits

Scientific research permits issued by the NMFS currently authorize studies of listed species in the Pacific and Atlantic Oceans which occur primarily in the action area. Tables 38 identifies the cumulative number of takes for each listed marine mammal and sea turtle species, as well as smalltooth sawfish, rockfishes, sturgeon, and eulachon in the action area authorized in scientific research permits. Due to the large numbers of research permits authorizing salmonid takes, we did not quantitatively assess the numbers of actions on listed salmon and steelhead. We did, however, address these in a qualitative manner. Cetacean takes include approach, biopsy, suction cup and implantable tagging, breath sampling, acoustic playbacks, and/or ultrasound. Pinniped takes consist of approach, capture, handling, restraint, measurements, anaesthesia or drugging, temporary captivity, external tagging, translocation, mortality, swab sampling, blood and tissue sampling, lavage, hot branding, tooth extraction, and/or ultrasound. Sea turtle researcher involves approach, capture, handling, restraint, PIT, flipper, satellite, or sonic tagging, lavage, mortality, ultrasound, blood or tissue sampling, captive experiments, laproscopy, imaging, and/or antibiotic injections. Smalltooth sawfish may be captured via a variety of means, measured, tagged, tissue sampled, and/or ultrasounded. Research actions on sturgeon species include capture, handling, restraint, anaesthesia, laproscopy, lavage, boroscopy, fin, operculum, or barbel clipping, PIT, floy, sonic, or satellite tagging, gonad sampling, prophylactic, and/or mortality. Rockfishes may undergo capture, handling, restraint, anaestheia, floy or sonic tagging, and mortality.

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

Table 38. Authorized takes of non-salmonid listed species in the action area.

Species	2009-2013 lethal take	2009-2013 sub lethal take
Humpback whale-North Atlantic	0	42,196
Humpback whale-North Pacific	0	340,137
North Atlantic right whale	0	31,805
Cook Inlet beluga whale	0	41,769
Southern resident killer whale	0	23,458
Hawaiian monk seal	59	41,590
Steller sea lion-eastern DPS	908	1,363,064
Green sea turtles-North Atlantic	71	69,045
Green sea turtles-North Pacific	0	12,490
Hawksbill sea turtle-North Atlantic	18	25,334
Hawksbill sea turtle-North Pacific	0	1,500
Kemp’s ridley sea turtle	34	31,865
Leatherback sea turtle-North Atlantic	12	18,759
Leatherback sea turtle-North Pacific	0	2,957
Loggerhead sea turtle-North Atlantic	402	92,385
Loggerhead sea turtle-North Pacific	0	1,886
Olive ridley sea turtle- North Atlantic	5	3,535
Olive ridley sea turtle-North Pacific	0	7,686
Smalltooth sawfish	0	3,940
Shortnose sturgeon	34,794	101,280
Green sturgeon-southern DPS	927	15,755
Bocaccio	155	191
Canary rockfish	2,919	3,182
Yelloweye rockfish	397	476

Permit numbers: 1420, 1440, 1447, 1449, 1450, 1462, 1475, 1486, 1501, 1505, 1506, 1507, 1514, 1516, 1518, 1522, 1526, 1527, 1537, 1538, 1540, 1542, 1544, 1547, 1549, 1551, 1552, 1556, 1570, 1571, 1575, 1576, 1578, 1580, 1581, 1591, 1596, 1599, 10014, 10018, 10022, 10027, 10037, 10042, 10045, 10115, 10137, 13306, 13307, 13430, 13543, 13544, 13573, 13675, 13815, 13846, 13908, 13927, 13958, 14073, 14097, 14122, 14176, 14204, 14210, 14233, 14245, 14249, 14266, 14272, 14274, 14296, 14325, 14326, 14336, 14337, 14353, 14381, 14390, 14394, 14438, 14442, 14451, 14457, 14462, 14477, 14506, 14508, 14510, 14534, 14585, 14586, 14591, 14599, 14603, 14604, 14616, 14622, 14655, 14682, 14726, 14759, 14780, 14791, 14824, 14949, 14971, 15112, 15119, 15135, 15225, 15237, 15249, 15269, 15271, 15274, 15275, 15330, 15372, 15408, 15415, 15431, 15449, 15476, 15483, 15488, 15545, 15552, 15566, 15595, 15606, 15614, 15677, 15682, 15741, 15750, 15783, 15791, 15801, 15847, 15849, 15883, 15994, 16017, 16039, 16073, 16146, 16166, 16174, 16194, 16208, 16220, 16262, 16269, 16271, 16292, 16327, 16328, 16347, 16408, 16414, 16429, 16439, 16458, 0642-1536, 0662-1661, 1036-1744, 1049-1718, 1058-1733, 1071-1770, 1120-1898, 1121-1900, 1127-1921, 1128-1922, 1410-4M, 14457-2M, 14457-3A, 15119-2A, 1586-3R, 473-1700, 532-1822, 540-1811, 545-1761, 587-1767, 594-1759, 605-1904, 633-1763, 633-1778, 715-1885, 716-1705, 727-1915, 731-1774, 753-1599, 774-1714, 775-1875, 781-1824, 782-1719, 808-1753, 945-1776, 948-1692, 965-1821, and 981-1707.

EFFECTS OF THE PROPOSED ACTION:
EFFECTS OF THE NUMERICAL STANDADRD

Exposure Analysis

In this section of the Opinion, we assess the probable direct and indirect effects to listed species and critical habitat that are reasonably likely to occur as a result of activities authorized, funded, or carried out by the ballast water management program including the recently proposed numerical standard. As discussed in the approach to the assessment, in order to determine the probability and extent of exposure, we rely on the spatial and temporal occurrence of the species identified in the *Status of the Species* section of this Opinion and the extent of the impacts anticipated in the *Action Area* section. The specific stressors associated with ballast water intake and discharge are capture of listed species during uptake of ballast water and the introduction of non-native species through ballast water discharge at the proposed standard. NMFS does not expect any effects from three methods of ballast water maintenance identified in the FR (33 CFR 151.1510 and 33 CFR 151.2025): (1) retaining ballast water on board, (2) discharging ballast water to on-shore or vessel/barge based holding facilities, or (3) using public drinking water, meeting the requirements of the Safe Drinking Water Act (42 U.S.C. §§ 300f-300j), as ballast water.

Ballast Intake

Global shipping moves 80% of the world's commodities (Tamburri *et al.* 2002, Bax 2003). When ships arrive in waters of the US to unload cargo, the ships take on ballast water to offset the cargo weight being removed. When these ships take on or discharge ballast, water enters or leaves the ship through two to six sea chests (depending on the size of the vessel) along the sides or bottom of the ship. When ships take on ballast, they are docked at ports and stationary. In the past, during ballast water exchange, they would take on ballast water while underway at off-shore sites, but under the USCG rule, ballast water exchange is no longer required. Ballast intake occurring at the same time and place as threatened or endangered species poses the potential for listed species to be harmed by impingement, entrainment, or capture in ballast tanks or pipes.

Impingement occurs when organisms that are sufficiently large to avoid passing through screens are trapped against the screen by the force of flowing water (for example, see Hanson *et al*. 1977). Entrainment occurs when organisms are drawn into an intake system with the water flow (for example, see Beck and Miller 1974, Hanson *et al*. 1977). Impingement typically involves adult fish and invertebrates and often results in death, physical damage to scales and dermal tissues, or physiological stress responses with stress pathologies (for example, see Brett 1958). In contrast, entrainment typically involves small organisms (such as invasive species), eggs, and fish larvae.

In most cases, large marine mammals such as whales will not be present in ports and will not be subjected to ballast water intake operations. Even in cases when smaller marine mammals such as steller sea lions may be present near ships during ballast intake, these animals are too large to get impinged or entrained on pipes or screens leading into the sea chest and strong enough to swim out of the area. Therefore, marine mammals under NMFS jurisdiction will not be affected by ballast water intake operations.

Sea turtles, like marine mammals, attain large sizes; however they comparatively may be more vulnerable to ballast water intake pipes. Sea turtles, especially recently hatched turtles, could be sucked into sea chests during ballast water intake activities. Recently hatched sea turtles leave their nesting beaches and swim to offshore rearing areas near debris collecting at convergence zones. It is unlikely that a hatchling sea turtle would be in a port because ports tend to be in harbors or upstream in estuaries away from beach environments. Adult sea turtles would also not be expected to enter ports, but if they did, they are powerful enough swimmers to avoid being impinged or entrained in sea chests or pipes during ballast water intake. Therefore, sea turtles may be affected but are not likely to be adversely affected by ballast water intake.

In the event that juvenile Atlantic or Pacific salmonids or shortnose, Atlantic, Gulf, or green sturgeon are in close proximity to ships taking up ballast water, these fish are not likely to get impinged or taken up. Based on a report on ballast water exchange prepared by the American Bureau of Shipping (2010), a moderately-sized ballast intake system would intake less than 250 cubic meters per hour (or about 0.069 m^3/s). Flow rates through sea chests are calculated to be less than 0.6 meters per second (m/s) (NMFS 2004), which would be the equivalent of a pipe that is 10 centimeters in diameter in the scenario presented above, but normal intake pipes are $1.1m^2$. Fish must be able to swim faster than 0.6 m/s or not occur in the same time or place as the ballast intake to avoid being impinged, entrained, or captured. Salmon can swim at speeds ranging from two to eight m/s (Taylor and McPhail 1985, Colavecchia *et al.* 1998) (Booth *et al.* 1997) and sturgeon have sustained swimming speeds above 1m/s and burst speeds over 4m/s (Adams *et al.* 1999, Allen *et al.* 2006, Webber *et al.* 2007). Therefore, salmon and sturgeon would be able to swim away from intake structures. It is possible that these species wouldn't avoid the intake current, however, adults of both species would be too large to be affected. Juvenile sturgeon, while very small, move to the deepest parts of main channels where they feed and grow. Juvenile salmonids rear in headwater systems as small juveniles and move to larger estuaries for parts of their lives as large juveniles or smolts. In both cases, interactions with vessels would be limited if they ever occurred and the most vulnerable sizes would not be exposed to ballast water intake systems.

Smalltooth sawfish occupy shallow nursery areas such as inshore bars, mangrove swamps, and sea grass beds as juveniles and then move offshore as adults. Ships would not be expected to uptake ballast water in these shallow areas, so interaction with juvenile smalltooth sawfish would be rare. If ships attempted to intake ballast water in a location with juvenile smalltooth sawfish, their uptake would be unlikely due to their large size at birth (very small juveniles are considered to be less than one meter, Wiley and Simpfendorfer 2010). Adults can grow to six meters in length (Simpfendorfer *et al.* 2008) and would not be vulnerable to uptake in ballast water tanks.

Rockfish are unlikely to be affected by ballast water intake because their habitat does not overlap with shipping ports. Rockfish adults live at depths between 50 and 250 meters (Gunderson and Sample 1980). Larval and juvenile rockfish are found near the thermocline in open ocean waters (Lenarz and Echeverria 1991).

Pacific eulachon, like rockfish, spend most of their lives in deep waters off the Pacific coast. However, Pacific eulachon are anadromous and return to freshwater to spawn before dying (Emmett *et al.* 1991a, Musick *et al.* 2000b, WDFW and ODFW 2001). During this brief time when adults enter freshwater during late winter and early spring and larvae are carried downstream to the ocean in spring and early summer, there is the potential for some interactions with vessels uptaking ballast water. Adults, because of their strong swimming ability, are expected to avoid ballast water intake pipes. However, larvae moving downstream would be vulnerable to ballast water intake for approximately three weeks until they are large enough to move to deep water areas in the river (Howell 2001). There are seven large ports in the lower Columbia River, where interactions between out-migrating larval Pacific eulachon and ballast intake pipes are possible. The currents in the Columbia River are unlikely to carry larval eulachon into them to make them vulnerable to ballast uptake, but there is a negligible chance during approximately a five week time in spring/early summer that some eulachon larvae could be removed from the system by ballast uptake.

There are several invertebrates under NMFS jurisdiction that, like Pacific eulachon, produce larval offspring which are passively carried by ocean currents. The black and white abalones live offshore in the Pacific Ocean and their offspring that will successfully establish in offshore rocky habitats would be unlikely to have any interaction with vessels intaking ballast water. Elkhorn and staghorn coral typically live in shallow water reef systems ranging from 1 to 10 meters deep and 5 to 30 meters deep, respectively (Wapnick *et al.* 2004, Mayor *et al.* 2006) (Gladfelter *et al.* 1977). Both species produce planktonic larvae. Neither would be expected in ports, which are generally dredged environments with sandy bottoms. However, these corals live near port environments and their offspring could be taken up during ballast water intake if currents have carried their offspring to a port location. Like with abalone species, any offspring that has drifted to a port environment would not successfully settle and establish a new coral body because that is unacceptable habitat. It is possible that the planktonic coral could have continued drifting to suitable habitat but the chances of a coral plankton being removed from a port environment that would have otherwise successfully settled is discountable.

Johnson's sea grass grows in bays, inlets, and lagoons. The grass typically occupies areas with turbid water and high tidal currents. In some cases, it may grow near ports or areas where ships could take up ballast water; however, as a rooted plant, there is essentially no chance it would be uprooted and removed from the environment. Furthermore, Johnson's sea grass is known to have female flowers, but no male flowers have been identified. Because sexual reproduction and potential offspring are unknown, ballast water intake would not be expected to remove any seeds from the environment.

Ballast Water Discharge

An often overlooked ballast water discharge issue is the presence of chemical pollutants in ballast water taken from locations which are relatively more contaminated than the water receiving the ballast discharge. The natural presence of chemical pollutants in ballast water discharge is not under the USCG's legal jurisdiction. The potential for exposure to ballast water chemical pollutants and the direct and indirect effects of such exposures will be analyzed in EPA's biological evaluation for the vessel discharge permit. However, the effects of any pollutant generated in association with ballast water treatment will be analyzed in tiered consultations to this programmatic because the types of treatments and their affects are unknown at this time.

In this Opinion, the only concern with ballast water discharge is the establishment of non-native species and how those non-native species may affect listed species and their critical habitat by altering the basic functional components of ecosystems. Several authors have argued that ballast water discharge in the global shipping industry represents a planetary scale homogenization of aquatic life due to the establishment of formerly isolated biota that are presently becoming mixed and effecting environmental change throughout marine and estuarine ecosystems (Lövei 1997, Ruiz *et al.* 1997, Lodge *et al.* 1998, Rahel 2000, Leppaekoski and Olenin 2001) (Carlton 1995). The ability of invasive species to alter population, community, and ecosystem structure and function is well documented (Elton 1958, Mooney and Drake 1986, Vitousek *et al.* 1987, Drake *et al.* 1989) and their direct and indirect effects to listed species and their critical habitat will be the focus of the effects analysis. Non-native species most often have indirect impacts to listed species through habitat alteration, which can result in changes in prey availability, changes in accessible habitat or cover, changes in predation risk due to effects on water clarity, and changes in water quality. Non-native species can also affect listed species or their critical habitat directly through competition, predation, or disease.

There are five groups of organisms that cause concern in ballast water: (1) planktonic life in suspension in the ballast tanks and ballasted cargo holds, (2) fouling organisms that settle on the lower walls of ballast tanks and ballasted cargo holds, (3) infaunal organisms living in benthic sediments of ballast tanks and ballasted cargo holds, (4) encysted stages and resting eggs of the above organismal groups in ballast tanks and ballasted cargo holds, and (5) fish and fish eggs in ballast tanks and ballasted cargo holds. The proposed ballast water standards focus on reducing planktonic life but may not fully address fouling organisms, infaunal organisms, or fish (USCG 2011). Because the BWDS only addresses discharge water, which would contain the suspended organisms, all potential introductions associated with ballast water are not addressed; however, treatment technologies and methods used to reduce the concentrations of planktonic organisms in ballast water should reduce members of other organismal groups.

In ballast water, microorganisms may constitute a numeric majority of the species found. (Galil and Hulsmann 1997, (Carlton 1999, McCarthy and Crowder 2000, Carlton 2001). Microorganisms, such as phytoplankton, particularly diatoms and dinoflagellates, can be especially abundant and may cause the harmful algal bloom known as “red tides” Hallegraeff 1993, NRC 1996) (Carlton and Geller 1993) or produce toxic compounds. While there was previously not a scientific, enforceable, or practical discharge standard for microorganisms, the USCG is now able to test ballast water and determine whether the ship being tested is meeting the proposed numerical standard.

Fish may not be as numerically abundant in ballast water as other phyla (Wonham *et al.* 2001), but once established in a new environment, fish can spread over broad geographic areas. A classic example is the Eurasian ruffe (*Gymnocephalus cernuus*), which was introduced via ballast water to the St. Lawrence River in the mid-1980s. Since its introduction, it has spread rapidly through rivers, bays, and lakes in the Great Lakes area. Furthermore, the ecological and geographical expansion of the ruffe has injured native fish populations (Anttila *et al.* 1998).

Susceptibility to Invasion

In areas with high species richness, non-native species are unlikely to be able to establish in a new environment because the native flora and fauna are better able to compete for and utilize the resources they have evolved to use (Stachowicz *et al.* 1999, Seabloom 2003) (Barko and Smart 1981). Increasing the number of reproductive members of an invasive population increases the probability that a successful establishment will occur (Kolar and Lodge 2001). The structure and function of aquatic and coastal ecosystems are adversely affected by increased stress from human activities, which have altered these systems for a long period of time. The scale, intensity, and rate of human activities and associated impacts have significantly increased in the past century from factors such as population growth, higher levels of consumption, and technological advances, making these systems more vulnerable to invasions by non-native species (Barko and Smart 1981, Cohen *et al.* 1984, Bogan 1993, Cohen and Carlton 1998, Maggs and Stegenga 1999, Ruiz *et al.* 1999, Cozzi *et al.* 2010, Strayer 2010, Norkko *et al.* 2011).

In addition to directly altered habitat, indirect habitat alterations such as climate change can contribute to the increased risks of invasions. Increasing winter water temperatures in the mid and high latitudes (Christensen *et al.* 2007) can provide more favorable conditions for invasive species to become established. Stachowitz *et al.* (2002) found that increased winter water temperatures were correlated with increased recruitment of non-native sea squirts and concluded that continued climatic warming could facilitate invasions of species from less extreme climates.

Due to their tolerance for a variety of salinity conditions and port locations in estuarine areas, estuarine or brackish water species are more likely to establish than species from other habitats (Nehring 2006a). Estuaries and coastlines are particularly susceptible to invasives because of their proximity to sources of invasives, such as ballast discharge (Carlton and Geller 1993, Ruiz *et al.* 2000, Williams and Grosholz 2008). Species capable of asexual reproduction may also be better able to establish in new habitats due to the lack of allee affects, bottlenecks, and other genetic effects associated with sexual reproducers (Roman and Darling 2007).

Progression of an Invasion

When assessing when a listed species or its critical habitat may be affected by a non-native species, the most important question is when in the invasion process can listed species or their critical habitat be affected? Carlton (1985), Baldwin (1992), Hallegraff (1998), and Carlton (2005) identified seven steps in the ballast mediated invasion process. The first step requires the potentially invasive species to be present at that time of year or under those water quality conditions creating the potential for an interaction between that species and the vessel. Second, if the potential invasive species is present at the port with the ship, it must then be taken up during ballast water intake, but also survive being taken up into the ballast tanks. Now in the ballast tanks, that individual or those individuals of the potentially invasive species must survive during transport from that foreign port to a US port. Fourth, upon arrival in the US, the potentially invasive species must be in the discharged ballast water and survive the discharge process. Fifth, upon release, the potentially invasive species must survive in the new environment. Sixth, the newly introduced species must establish a reproductive population, producing viable offspring. And finally, after that species is able to reproduce, it must survive and reproduce effectively enough to establish a long-term population in the area.

While the likelihood that any given individual of a particular species might go through all the steps previously described and become successfully established in the long-term is quite low, the likelihood that a vessel will release live organisms from ballast water (stage five) is very high, especially given the large volumes of ballast water discharged annually in US ports. It is possible that listed species or critical habitat exposed to recently released organisms could exhibit a response, but the potential affects to listed species and their critical habitat increase with the duration of exposure (stage six and seven) and the wider the exposed area. Therefore, exposure to listed species is considered to have occurred when reproduction occurs in US waters and viable offspring are created (stage six).

The contention in the final PEIS is non-indigenous species “management is best accomplished by preventing introduction and establishment, which translates into minimizing release. Setting a discharge standard may reduce release and thus decrease the probability of introduction and establishment” (p. 4-6, 4-7, Davis 2005). While, as previously discussed, if no organisms are present in the ballast water discharge or if no ballast water is discharged into the environment, there is no invasive species effect to the receiving environment. However, most vessels will have initial organism densities greater than the organism densities proposed by the USCG so that vessels meeting any of the alternatives will achieve a general reduction in the non-native species densities. Therefore, if ballast water discharge is going to have non-native species present, and indeed, globally between 3,000 and 7,000 individual species are likely being transported in ballast tanks (Carlton and Geller 1993) (Dunston and Bax 2003), the level of inoculation is critical to the risks of non-native species establishing viable reproductive populations (Bax 2003).

For transoceanic introductions, vessel-mediated transport is virtually the only vector, but organisms may arrive in the US via several vectors, including the hull, anchor system, sea chest, internal seawater piping system, and the bilges (Keller and Lodge 2007) (Padilla and Williams 2004, Coutts and Dodgshun 2007). Furthermore, non-native species establish in US waters from other vectors besides vessels such as, birds (Johnson and Carlton 1996, Green and Figuerola 2005), aquaculture (Padilla and Williams 2004), Keller and Lodge 2007), aquarium trade (Padilla and Williams 2004), Keller and Lodge 2007), and live bait (Johnson and Carlton 1996). Nonetheless, the comparatively large volume of ballast water and its protection from open-ocean conditions during transit implies that ballast water discharge is likely to be a primary source of non-native species associated with shipping. The reductions of organisms released via ballast water discharge is dependent on the complex interplay of a number of factors: the types of organisms; their reproductive mode; their life stage; their nutritional status; the physical match between source and receiving waters; the time of the year; suitable substrate (if the species has a benthic phase); the presence of other invaders that might facilitate establishment; presence of native or non-native competitors, predators, parasites, or diseases that might inhibit establishment; pollutants in the receiving waters; and frequency of release of a given organism.

Invasion rates vary considerably with region and propagule delivery. One of the main factors to be considered is the large variability in the densities (concentrations) of organisms transported via ballast water. Sampling studies show that organism densities in a ballast tank or ballasted cargo hold can range from as little as one individual per cubic meter (very low densities) to thousands per cubic meter (very high densities) (Cohen 1998, (Carlton 2001). These studies rarely focus on the bacterial and viral components of ballast, but these organisms may exist in even greater densities. The greater the volume discharged the more non-native species that can be introduced to the local environment. Additionally, multiple ballast discharges in the same port from similar source waters likely contain many of the same species, increasing the invasion risk. As Minton *et al.* (2005) noted, "Even with the more stringent IMO standards, however, per-ship discharges in excess of 10^6 total zooplankton remain possible, as a result of the large ballast discharge volumes of some ships. Moreover, the size and frequency of ship discharges to individual port systems are additive in terms of propagule delivery, and may operate in concert to affect the risk of invasions." Non-native species from ballast water discharge are also additive to non-native species already established (stage 6 and 7) or recently discharged from another ballast tank or ballasted cargo hold (stage 5).

The amount of shipping traffic and the number of incidents of invasions per year are both increasing. In this way, viable populations can be artificially increased or small populations could become viable. Additive impacts from multiple ballast discharges can increase small, but non-invasive viable populations (stage 6) into problematic populations (stage 7) or increase small numbers of individuals (stage 5) into viable populations (stage 6) (Ricciardi 2001). Furthermore, in some cases, benign viable populations can also be allowed to grow to problematic sizes when a second introduced species competes with a native species that had been dominant to the initial non-native species (Ricciardi 2001). This was demonstrated in a study of native mussels, non-native clams, and non-native crabs, where the newly introduced crabs selectively preyed on the native mussels, thereby freeing the non-native clams (which had been living in the area for at least 50 years without having a population explosion) from competition and facilitating their spread in the ecosystem (Grosholz 2005).

When ballast water is discharged, there are generally many non-native species poured into the environment, but due to biotic and abiotic factors, those species rarely live beyond their introduction. While it is generally accepted that the most important vector responsible for transporting and introducing non-native species is ballast water discharge (Carlton and Geller 1993, Tamburri *et al.* 2002) USCG 2011), NMFS was only able to find one study that discussed the relative importance of various vectors. Of the 329 aquatic invasive species analyzed by Molnar *et al.* (2008), potential sources of those organisms were up to 69% via shipping, 41% by aquaculture, 17% by construction, 6% by aquarium trade, and 2% by live seafood trade. According to Molner *et al.* (2008), of those 69% (227 species) introduced by shipping, 39% were exclusively introduced by hull fouling, 31% exclusively by ballast water, and the other 30% by both.

Invasive Species History

Pre-Ballast Water Regulations

Invasion corridors have been open since ships first sailed from one port to another; however, non-native species generally did not establish in the US until early in the 19th century (Grosholz 2002) USCG 2011). As ships have increased in size, their ballast tanks are larger and capable of discharging a greater volume. Untreated ballast water has between 3,000 and 50,000 large organisms (over 50 microns) per cubic meter, resulting in a discharge of up to 100,000,000 large organisms and a national average of 10,000,000 large organisms per discharge (Minton *et al.* 2005). These discharges and resulting invasions have occurred at an increasing rate through the 1980s, 1990s, and 2000s.

Identifying the precise time of an invasion is often difficult because there is a lag between when the species are discharged (stage five) and when they are discovered (generally stage six or seven). To illustrate how complicated predicting the invasion process can be, Carlton (1996) noted that in some cases, invasion pathways have been open for decades with vessels moving between ports and discharging ballast water without a non-native species establishing, then for no explainable reason, a species like the zebra mussel becomes a problematic invader. While it may be impossible to know for sure why the zebra mussels established in the 1980s instead of the 1960s, increased habitat alteration has made many areas more susceptible to colonization by non-native species (Brothers and Spingarn 1992, Carlton 1996, Lake 2004, Didham *et al.* 2007). Furthermore, some non-native species that are currently in stage six of the invasion process may have been present for decades without being noticed because of a lack of environmental monitoring or their harmless nature.

Annually, reproductively viable populations of non-native species were being introduced at the rate of one species per year in the early 1800s, increasing to approximately 3 new species a year in San Francisco Bay alone between the 1960s and 1990s (Cohen and Carlton 1998). By the early 1990s, non-native species comprised 40 to 100% of the common species in the San Francisco Bay, 97% of the total organisms, and 99% of the biomass there (Cohen and Carleton 1995, Cohen and Carlton 1998). This story was similar throughout the US as habitat alteration and economic globalization led to increasing reports of ecological and economic impacts from invasive species. As a result of the social problems resulting from invasive species introductions, Congress passed NANPCA in 1990 and amended the law in 1996 when they passed NISA in an effort to control the rate of invasions.

Ballast Water Exchange

Ballast water exchange, when ocean water from outside the EEZ is pumped through the ballast tanks to exchange the ballast water taken up from nearshore port environments and replaced with less species-rich water from the mid-ocean, was made mandatory for the great lakes in 1993. It was then made voluntary for ships entering all US waters in 2001. In 2004, BWE was made mandatory as a ballast water treatment method intended to protect US waters from invasive species and their impacts. The effectiveness of BWE varies dependent upon vessel type (design), exchange method, ballasting system configuration, exchange location, and method of study. One group of studies suggests that the efficacy of ballast water exchange is 80 to 99% per event (Rigby and Hallegraeff 1993, Smith *et al.* 1996, Dickman and Zhang 1999, Zhang and Dickman 1999, Hines and Ruiz 2000, Taylor and Bruce 2000). Other studies demonstrate that the volumetric efficiency of BWE ranges from 50 to 90% (Zhang and Dickman 1999, USCG 2001, Battelle 2003). Recent work by Ruiz *et al.* (in NOAA 2007) has shown that BWE can range from 88% efficacy of BW removal in container ships to between 97% and 99% in tanker ships, container ships, and bulk carriers. Locke *et al.* (1993) found a 67 to 87% exchange efficacy in removal of brackish-water-tolerant organisms from ballast water after mid-ocean exchange. Furthermore, studies using methlylene blue dye in ballast tanks have shown a 75 to 95% (for the phytoplankton community) efficacy was demonstrated in a test where ballast water was flushed through tanks for a continuous nine hours (equivalent to three tank volumes) (Wonham *et al.* 1996). Smith *et al.* (1996) showed a reduction in concentrations of larvae and plankton by 50 to 90% in ballast tanks after mid-ocean exchange. Ruiz *et al.* (in NOAA 2007) found that empty-refill BWE removed between 80% and 95% of zooplankton and 74% and 93% of phytoplankton (mean values from multiple samples). Dunston and Bax (2008) showed that exchanging ballast water three times, removed 95% of the organisms, but reduced the risk of invasion by only 21%.

Under BWE management, the reductions in organisms in ballast tanks resulted in reductions in the number of invasions from the highs seen in the 1990s. Mid-ocean BWE reduces the organism density by a factor of zero (upper bound) to one-tenth (lower bound) (USCG 2003). As noted above, without any treatment, the mean ballast water discharge contained 10,000,000 large organisms (range of 100,000 to over 100,000,000) and under BWE the number of introduced large organisms fell to a mean of 1,000,000 (Minton *et al.* 2005). The reduction in organisms corresponded to an annual reduction in the number of established, reproductively viable, non-native populations by 53.7 to 62.7%. However, due to the variable success of BWE and some vessels not being able to conduct BWE because of safety issues, the USCG decided to establish a numerical standard to further reduce the potential for establishing invasive species. Under the proposed management program, BWE will no longer be an authorized treatment method and would only be used by vessels entering the Great Lakes.

Despite the dramatic reductions in the number of organisms released, by each ballast discharge, the number of organisms being released in small geographic locations is very high. In the Port of Hampton Roads in the Chesapeake Bay, Drake *et al.* (2007) estimated that 10^{20} microorganisms, 6.8×10^{19} viruses, and 3.9×10^{18} bacteria cells are discharged via ballast water annually. Lawrence (2008) estimated 3×10^{22} viruses to be transported annually. In 2005, a total of roughly 22 million tons of ballast water was discharged at the four largest west coast ports; 27% of this ballast water originated from one of the other three major US west coast ports (Simkanin *et al.* 2009). Of the taxa assessed in these discharges, Drake *et al.* (2007) estimated that 56% could survive in the discharge habitat. Egg, larvae, and juvenile life stages appear to be the most common life stages of animals and plants establishing invasive populations (Mahtab *et al.* 2005, Von Halle and Simberloff 2005)).

Proposed USCG Numerical Standard

The goal of the numerical standard is to further reduce the number of organisms released in ballast water from BWE and to also increase the total number of vessels that treat ballast water. While BWE results in the discharge of 1,000,000 large organisms, under the worst possible conditions, the numerical standard would release that many large organisms, but on average, the number of large organisms discharged would be reduced to 43,500 (Minton *et al.* 2005). And while research on the number of smaller organisms discharged under BWE hasn't been conducted, the largest expected discharge of small organisms is 100,000,000,000 and the mean number of small organisms that would be discharged is 4,350,000,000 under the numerical standard.

BWE is more effective than numerical standards for freshwater when ballast water is discharged due to the salinity differences (Gray *et al.* 2007). Despite the loss of the lone potential benefit of BWE, this numerical standard will reduce the number of organisms per volume from the current BWE levels. To assess the impacts of establishing this new standard, it is necessary to calculate the likelihood of a new non-native population establishing. Furthermore, NMFS must consider whether more stringent standards would provide additional protection for listed species and their critical habitat. NMFS uses analyzed the probable exposure associated with a standard 1000 times more stringent than the currently proposed standard because that is the alternative identified in the final PEIS, the Notice of Proposed Rule Making, and IFR.

The final PEIS uses a population viability analysis (PVA) to relate initial population size and extinction probability to determine the probability of successful establishment of new non-native populations. A PVA is normally used to determine the size of a native, viable population necessary to ensure a high probability of survival. In the context of this final PEIS, the concern is the initial non-native species population size must be small enough to ensure a high probability of extinction. While PVAs are intended for native species relying on native resources, NMFS is unaware of a better model and furthermore, theoretical ecology suggests that stochastic events and allee effects are more likely to cause the extinction of small populations, regardless of whether they are native or non-native. Therefore, NMFS is unaware of a better model for predicting the likely changes in invasion risk from BWE to a numerical standard.

The PVA analysis in this final PEIS is based on the diffusion model proposed for the analysis of population viability by Dennis *et al.* (1991). The model involves two main parameters, one relating to population growth rate and the other to population variability around mean growth. In conjunction with an estimate of the proportional reduction in initial population size resulting from the alternatives, this model allows for a comparison of the alternatives in terms of their effects on the probability of reduced introductions of organisms.

The diffusion model provides an assessment of the *relative, proportional* effectiveness in increasing extinction probability of a particular ballast water standard as compared to untreated ballast water or BWE. The USCG did not perform a comparison in *absolute* terms because it would be very difficult and the results would be highly uncertain given the scientific knowledge of invasive species ecology when this analysis first began in 2001 (Ludwig 1999).

The main premise of the model is that a decrease in the number of living organisms initially introduced reduces the probability that a population becomes successfully established. This is because there is a relationship between initial population size of the introduced organisms and their extinction probability (Colautti *et al.* 2006) (Hopper and Roush 1993, Memmot *et al.* 1998, Grevstad 1999)(Colautti *et al.* 2006). While this model takes into account allee effects and reproductive events (stage six) of the newly introduced species, it fails to account for supplemental ballast mediated introductions of the same species to the area from discharges from other vessels (Dunstan and Bax 2008). Introduction of individuals in separate events can increase the genetic diversity available for an invasive species, allowing for more rapid evolution and exploitation of the new environment; the same can occur from high levels of variation within an establishing population (Roman 2006, Williams and Grosholz 2008). In the late 1990s, 21 billion gallons of ballast were discharged into ports of the US (Bright 1999, EPA 2001). By 2010, over 64 billion gallons of ballast water were being discharged by ships discharging between 1,500 and 103,000 cubic meters at a time (Minton *et al.* 2005). Despite increased precautions associated with NISA, new introductions are occurring throughout the United States at an increasing rate as a result of increased ship traffic (Grosholz 2005). Therefore the PVA model used by the USCG likely over-estimates the extinction likelihood of non-native species and therefore, the effectiveness of the numerical standard.

The PVA modeled by the USCG provided a range of possible effectiveness using models that assume either a single species discharged from ballast tanks at the numerical standrad levels or multiple species discharged from ballast tanks at numerical standard levels. The single species model reported up to an 8% reduction in the introduction rate of small organisms, but between 0 and 50% reduction in the likely introduction of large organisms. The long-term objective was also modeled, which would result in a 0 to 33% reduction of small organisms and a 0 to 59% reduction of large organisms. However, this model is not ecologically realistic because it is unlikely that every organism discharged by ballast water is the same species. The multiple species scenarios presented in the final PEIS are more ecologically realistic than evaluating a single species scenario. Under the multiple species scenarios, the numerical standard could reduce the chance of invasion by between 37- and 63% from the invasion risk present in BWE. However, because the PVA model likely overestimates the effectiveness of the numerical standard by not considering the volume of ballast water from multiple ships, NMFS chose to use the 37% reduction in risk to be conservative and protective of listed species. Following the same rationale, the long-term objective of the ballast water program could be conservatively 85% more effective than BWE.

Estimating Future Invasion Rates

Many studies have been conducted to determine a valid model for predicting invasion rates, including recent assessments assessing the proposed numerical standard performed by the National Academy of Sciences (2012) and EPA (2011). The goal of both of those studies was to predict the invasion probability of individual species given propagule pressure. The two studies determined there are no valid models to address the questions they were tasked with answering. Both of these reviews are complete and address the limitations of our knowledge of invasive species modeling to date. NMFS agrees no models are infallible and that the invasive species science is not well developed enough to predict specific invasive species probability of invasion; however, for the purposes of this national level programmatic, a more general model is used.

Drake and Lodge (2004) and Drake *et al.* (2005) provide a global estimate for the probability of an invasive species establishing given the volume of ballast water discharged. The goal of their model is "not to predict establishment probabilities of any particular species, but to predict invasion rates over a range of species" (NRC 2012, p. 105). The National Academy of Sciences goes on to say that, "the chance of a single invasion by a size class of organisms can be predicted as a function of the number and volume of releases, independent of the number of individual organisms" (NRC 2012, p. 105). The shortcoming of this model, as noted by the National Academy of Science is that it has not been validated.

With the intention of attempting to validate the global probability analysis provided by Drake and Lodge (2004) and Drake et al. 2005, NMFS estimated the probability of an invasion using their model as well as a validation technique using historic information from a well studied location. Applying the best scientific information available, the validation method requires knowledge of historic invasions for specific ports and a conversion from no ballast water treatment to the current ballast water standard to estimate the current invasion rate. This method presents two shortcomings: (1) there is limited information on historic invasion rates, with only sufficient knowledge from San Francisco Bay, and (2) like the USCG PVA analysis, it does not consider changes in ballast water discharge volumes or vessel traffic through time.

In our analysis, we rely on the PVA presented in the USCG final PEIS to estimate the anticipated invasion rate because it is currently the best available estimate of the impacts of implementing the numerical standard. Because the ballast water standard establishes a numerical limit of organisms of a particular size allowed in a specific volume of ballast water, ideally a predictive model would assess the probability of an invasion given the volume of ballast water discharged in a bioregion or the entire US in order to establish the anticipated annual invasion rate.

The PVA designed by the USCG provides information on the generally anticipated results of the numerical standard relative to the results of BWE. It indicates that initially at the numerical standard, assuming multiple species are discharged by ballast water, that there is conservatively a 37% proportional expected reduction in the number of invasive species introduced. Also, Dunstan and Bax (2008) show BWE reduces the likelihood of an invasive species establishing by between 43 and 48%. However, this doesn’t predict the absolute number of new non-native species that will be introduced each year, which is the number actually needed to estimate the likelihood that threatened or endangered species and their critical habitat may be exposed to non-native species.

The method of predicting invasive species based on their historic rate of introductions can only be done using San Francisco Bay; however because San Francisco Bay is the most invaded body of water in the US and possibly the world (Cohen and Carleton 1995, Connor *et al.* 2002, Costello *et al.* 2007), that invasion rate can be used as a conservative estimate for all ports in the US. Cohen and Carlton (1998) argue that while it is difficult to determine whether a newly discovered invasive species was recently established or had been present for a long time, all evidence suggests the rate of invasions was increasing in exponential fashion between 1850 and 1995. By the mid 1990s, invasions by non-native species were occurring at their greatest rate in over 200 years of introductions (Cohen and Carlton 1998). The NMFS analysis uses the estimated number of invasive species introduced in 1995, and the rate of introductions under BWE to estimate the expected rate of introductions using the PVA provided in the final PEIS. In all cases, the best available numbers used in the analysis were highly variable. Therefore, the numbers used for the estimates below represent the most conservative calculations by always using the upper error bounds of the most probable estimate or upper end of a range. NMFS acknowledges there is insufficient knowledge to get accurate numbers, but believes the estimated rates of invasion presented below represent the most conservative estimate.

In San Francisco Bay research first began on invasive species in 1851. At that time approximately one new non-native species would establish in the bay each year (Cohen and Carlton 1998). As shipping traffic increased, the number of non-native species that established in San Francisco Bay increased to 2.3 each year between 1961 and 1995 (Callaway and Josselyn 1992, Cohen and Carlton 1998). However, there are several sources of non-native species, and vessels account for approximately 69% of the time (Molner *et al.* 2008) (2.3*0.69=1.587 species introduced by vessels), of which as many as 61% of the organisms that establish from vessel mediated sources come from ballast water (1.587*0.61=0.968 introduced by ballast water) (Molner *et al.* 2008). Using this information, we estimate that by 1995, approximately 0.97 individuals introduced to the San Francisco Bay area each year were from ballast water. Dunstan and Bax (2008) showed that BWE reduced the probability of an established population of non-native species by 43 to 48%. This correlates with annual ballast-mediated non-native species introductions being reduced from 0.968 per year to approximately 0.504 (0.968*0.52) to 0.553 (0.968*0.57) per year. Then, using the multispecies analysis in the final PEIS (0.504*0.63 and 0.553*0.63), the proposed numeric standard would be expected to reduce the number of invasive species to approximately 0.318 to 0.348, or approximately one new species every three years.

The calculated rate of 0.318 to 0.348 invasions per year is comparable to the global multi species model used by NMFS to estimate invasion risk at the continental scale (Drake and Lodge (2004). Drake and Lodge (2004)(, p.578) used a generalized linear regression with Poisson errors developed at a global scale to show “that shipping traffic was significantly related to the number of nonindigenous species discovered ($y = 8.47 \times 10^{-8}x$ where y is the number of non-indigenous species discovered and x is the shipping volume, in metric tonnes; $p \leq 0.0001$).” Ballast water discharge volumes were obtained from the National Ballast Information Clearinghouse, which is a record of all reported ballast water discharges. However, reporting rates vary between 80 and 90% of the actual value. NMFS applied a correction factor assuming the conservative 80% reporting rate. This rate is expected to increase with the ballast water standard because numerical values are easier to test and therefore there is greater risk in not complying with the mandatory reporting requirements. In San Francisco Bay the corrected average from 2005 to 2010 was 6.469×10^6 tonnes of ballast water were discharged under BWE management. Using Drake and Lodge’s (2004) equation would result in approximately 0.548 species per year being introduced. Then, using the multispecies analysis in the final PEIS, which conservatively estimates a 37% reduction in new invasions, the expected number of invasions in SF Bay would be approximately 0.345 individuals per year (without the correction factor, the estimate is 0.32 species introduced per year).

While this test is not an exhaustive validation, both models in San Francisco Bay determine nearly identical estimates. Given the similarity of their estimates and the flexibility of the second method using the ballast water discharge volume at various ports, we believe this method provides the best available estimate for future invasions at individual ports as well as throughout the US.

Despite setting a ballast water standard, listed species and habitat will likely still be exposed to ballast water discharges with non-native organisms at any standard other than zero discharge of non-natives. If the volume of ballast discharge remains at the same level and there are no additional habitat alterations making the environment more susceptible to invasions, ballast-mediated invasions are likely to be reduced under the proposed numerical standard. However, changes in the volume of shipping and in trade routes, changes in habitat health and habitat accessibility, as well as changes in ballast water systems and their operation, would either decrease or increase the level of non-native species introductions. Thus, depending on factors outside the scope of the ballast water management program, the total number of introductions could still increase due to increases in global trade and habitat accessibility.

Case Studies

As was just discussed, San Francisco Bay is a logical location to use as a case study because of the vast research history as well as invasion history there. Furthermore, the San Francisco Bay is home to six listed species and five critical habitats in two different functional groups under NMFS jurisdiction. NMFS also decided to analyze Puget Sound and Chesapeake Bay to assess the impacts of invasive species on listed species and their critical habitat. By choosing those locations, several listed species, their critical habitats, or reasonable surrogates were not considered, so we analyzed those individually.

San Francisco Bay. Among the best studied port in the US is San Francisco Bay, with major ports in both San Francisco and Oakland as well as many smaller ports in what may be the most heavily developed large estuary in the US (Herborg *et al.* 2007). San Francisco Bay is also home to central valley spring-run Chinook salmon, Sacramento River winter-run Chinook salmon, green sturgeon, and California central valley steelhead (Table 39). Additionally, while they do not reside within the bay, black and white abalone live along the California Coast within the same marine bioregion.

Carlton and Geller (1993), Kennedy (2001), and the final PEIS estimate approximately 234 invasive species are now established in the estuary. Many of these species were established before any ballast water treatment was utilized. And as noted, there is considerable uncertainty as to when a species establishes due to the delay between it being discharged and it being discovered.

Table 39: Listed species present and critical habitat designated in San Francisco Bay.

Species	Listing Status	Designated Critical Habitat
Central valley spring-run Chinook salmon	Threatened	Yes
Sacramento River winter-run Chinook salmon	Endangered	Yes
Green sturgeon	Threatened	Yes
California central valley steelhead	Threatened	Yes
Black abalone	Endangered	Yes
White abalone	Endangered	No

The reported ballast water discharge into the San Francisco Bay bioregion has increased between 2005 and 2010 with 4.43×10^6, 4.6×10^6, 4.49×10^6, 5.9×10^6, 5.7×10^6, and 5.9×10^6 tonnes discharged annually from around the globe, respectively. These numbers were corrected for the unreported discharges. If we accept the USCG's PVA results, future invasions in San Francisco Bay at the discharge rate averaged over 2005 to 2010, would be approximately one new species establishing every three years under the proposed numeric ballast discharge standard. Following this trend in discharge volumes; however, an increase of approximately 372,000 tonnes per year in San Francisco Bay would be expected each year beyond 2010, increasing the risk of an invasion by approximately 0.032 new non-native species each year. In other words, if the rate of 0.345 new species per year in 2010 is correct, given the upward trend in ballast discharge provided by the National Ballast Information Clearinghouse, the potential for an invasive species to establish in 2012 would be approximately 0.407 new non-native species each year. On the other hand, if the USCG ballast water management program increases the standard in the event better technology is practicable, their goal is a standard that is 1000 times more stringent than the proposed numeric standard. In that case, the anticipated rate of invasions would fall to about 0.09 individuals per year or about one new non-native species every 11 years. Although we can estimate the rate of invasion within a given system, we cannot predict the species that would constitute future invasions.

Puget Sound. Puget Sound has a number of ports, including Seattle, one of the largest ports in the US (Herborg *et al.* 2007). Puget Sound is home to bocaccio, canary rockfish, Puget Sound Chinook salmon, Hood Canal summer-run chum salmon, Pacific eulachon, Puget Sound steelhead, and yellow-eye rockfish (Table 40).

Table 40: Listed species present and critical habitat designated in Puget Sound.

Species	Listing Status	Designated Critical Habitat
Bocaccio	Endangered	No
Yellow-eye rockfish	Threatened	No
Canary rockfish	Threatened	No
Pacific eulachon	Threatened	Yes
Puget Sound steelhead	Threatened	No
Puget Sound Chinook salmon	Threatened	Yes
Hood Canal summer-run chum salmon	Threatened	Yes
Pacific eulachon	Threatened	Yes
Resident killer whale	Endangered	Yes
Humpback whale	Threatened	No
Eastern Steller sea lion	Threatened	Yes

The number of non-native species in Puget Sound has increased dramatically over the past 30 years. The first analysis identified 18 invasive invertebrates (Carlton 1979). However, research nearly 20 years later by Elston (1997), Ruiz and Hines (1997), WDFW (1998), and Cohen *et al.* (1998) determined there were between 31 and 67 non-native species reproducing in Puget Sound. Cohen (2004) later re-sampled the sound and identified 76 non-native species.

Ballast water discharge into Puget Sound has increased between 2005 and 2010 with corrected volumes of 4.99×10^6, 5.16×10^6, 4.92×10^6, 6.12×10^6, 5.25×10^6, and 5.8×10^6 tonnes discharged annually from around the globe, respectively. The mean corrected discharge rate over this time was 5.37×10^6. Again, if NMFS adopts the USCG's PVA from the final PEIS, we would expect a 37% reduction in the number of new non-native species under the numerical standard and an 85% reduction under the ballast management program's published goal. Therefore, over the past six years, an invasion rate of approximately 0.455 non-native species per year (8.47×10^{-8} x 5.37×10^6 x 0.63) would be expected or approximately one new species in Puget Sound every two years. And as we did for San Francisco Bay, observing the increasing trend in discharge volumes, there is an increase of approximately 161,000 tonnes per year each year beyond 2010, increasing the risk of an invasion by approximately 0.014 new non-native species each year. In other words, if the rate of 0.455 new species per year over the past six years is correct, given the upward trend in ballast discharge provided by the National Ballast Information Clearinghouse, the potential for an invasive species to establish in 2012 would be approximately 0.483 new non-native species each year. On the other hand, if the USCG ballast water management program increases the standard to 1000 times more stringent than the proposed numerical standard, the anticipated rate of invasions would fall to about 0.068 individuals per year or about one new non-native species every 15 years. Although we can estimate the rate of invasion within a given system, we cannot predict the species that would constitute future invasions.

Chesapeake Bay. The Chesapeake Bay is home to two of the largest ports in the US (Herborg *et al.* 2007) as well as a number of smaller ports that also receive ballast water discharge. The Chesapeake Bay is the focus of national attention in an effort to restore the largest estuary and one of the most productive fisheries in the US. At various times throughout the year, shortnose sturgeon, five distinct populations of Atlantic sturgeon, and green, hawksbill, leatherback, Kemp's Ridley, and Northwest Atlantic DPS of loggerhead sea turtles use the Chesapeake Bay as their home. (Table 41).

Table 41: Listed species present and critical habitat designated in Chesapeake Bay.

Species	Listing Status	Designated Critical Habitat
Shortnose sturgeon	Endangered	No
Atlantic sturgeon	Endangered and Threatened	No
Green sea turtle	Threatened	Yes
Hawksbill sea turtle	Endangered	Yes
Leatherback sea turtle	Endangered	Yes
Loggerhead sea turtle	Threatened	Yes
Kemp's ridley sea turtle	Endangered	No

Carlton (1996) assessed the non-native species in the Chesapeake Bay and found only 20 aquatic invasive species. Following this assessment, Smith *et al.* (1999a) noted there were relatively few invasive species in the Chesapeake Bay because of poor water quality conditions. Simultaneously, another study accounting for known invasive species as well as species of unknown origin was conducted and determined there were 196 exotic or cryptogenic species in the Chesapeake (Ruiz *et al.* 1999). It is possible the second study uncovered more non-native species by taking more samples or because efforts to improve water quality are allowing non-native species to capitalize on the nutrient rich waters and greatly disturbed habitat in the bay.

Ballast water discharge into Chesapeake Bay has increased between 2005 and 2010 with corrected volumes of 6.72×10^6, 8.0×10^6, 10.76×10^6, 16.63×10^6, 14.45×10^6 , and 21.06×10^6 tonnes discharged annually from around the globe, respectively. This is the greatest proportional increase in ballast water discharge in the US during this time period. Again, if NMFS adopts the USCG's PVA from the final PEIS, we would expect a 37% reduction in the number of new non-native species under the numerical standard and an 85% reduction under the ballast management program's published goal. Therefore, using the six year average from 2005 to 2010, an invasion rate of approximately 1.1 non-native species per year (8.47×10^{-8} x 12.94×10^6 x 0.63) would be expected or approximately three new species in Chesapeake Bay every three years. As we did for San Francisco Bay and Puget Sound, observing the increasing trend in discharge volumes, there is an increase of approximately 2,869,000 tonnes per year each year beyond 2010. This increase in ballast water discharge is likely to increase the risk of an invasion by approximately 0.24 new non-native species each year. In other words, if the rate of 1.1 new species per year is correct, given the upward trend in ballast discharge provided by the National Ballast Information Clearinghouse, the potential for an invasive species to establish in 2012 would be approximately

1.34 new non-native species each year. On the other hand, if the USCG ballast water management program increases the standard to 1000 times more stringent than the proposed numerical standard, the anticipated rate of invasions would fall to about 0.164 individuals per year or about one new non-native species every six years. Although we can estimate the rate of invasion within a given system, we cannot predict the species that would constitute future invasions.

Smalltooth Sawfish, Johnson's Sea Grass, Elkhorn and Staghorn Coral

Smalltooth sawfish, Johnson's sea grass, and elkhorn and staghorn coral occupy similar ranges around the southern tip of Florida. The range of these corals extends to Puerto Rico, while smalltooth sawfish may range between Florida and some Caribbean Islands. Johnson's sea grass is found in isolated bays, inlets, and lagoons along Florida's Atlantic Coast. While smalltooth sawfish, Johnson's sea grass, and the corals exist in the same marine bioregions as ports in Houston and New Orleans (ballast water bioregion CAR-I), they only use the far eastern areas in this marine bioregion and therefore discharges from Mobile Bay and west were considered unlikely to affect these species. Despite excluding some major ports, this area is still exposed to discharges in Tampa, Miami, and the Florida Keys.

In the waters of Florida alone, 23 non-indigenous aquatic plant species have become established (McCann *et al.* 1996). Approximately 50 non-indigenous aquatic species have established in the state, but not all from ballast water discharge (Courtenay 1997).

Ballast water discharge into coastal Florida and Caribbean waters has remained steady between 2006 and 2010 with corrected volumes of 14.43×10^6, 15.19×10^6, 15.52×10^6, 14.26×10^6, and 14.08×10^6 tonnes discharged annually from around the globe, respectively. If NMFS accepts the USCG's PVA from the final PEIS, we would expect a 37% reduction in the number of new non-native species under the numerical standard and an 85% reduction under the ballast management program's published goal. Therefore, over the last 5 years, an average invasion rate of approximately 0.784 non-native species per year (8.47×10^{-8} x 14.698×10^6 x 0.63) would be expected or approximately four new species along the southern Florida coast every three years. And as we did for the three case studies, observing the trend in discharge volumes, there is a decrease of approximately 70,000 tonnes per year each year beyond 2010, decreasing the risk of an invasion by approximately 0.006 new non-native species each year. In other words, if the rate of 0.784 new species per year is correct, given the downward trend in ballast discharge provided by the National Ballast Information Clearinghouse, the potential for an invasive species to establish in 2012 would be approximately 0.772 new non-native species each year. On the other hand, if the USCG ballast water management program increases the standard to 1000 times more stringent than the proposed numerical standard, the anticipated rate of invasions would fall to about 0.187 individuals per year or about one new non-native species every five years.

United States estimate

Shipping is one of the primary means that the US imports and exports goods to the rest of the world. There are three general types of aquatic ecosystems in the US: freshwater, estuarine, and saltwater, and all of those systems receive ballast water discharges and can be affected by introduced species. EPA estimates that approximately 75,000 large ballasting vessels use US waters, but not all of those ships are regulated by this FR. Additionally, approximately 6,000 new ships are built each year around the world and the size of the new ships are much larger than the ships currently used for international trade (Bax *et al.* 2003).

The USCG's final PEIS estimates approximately 150 non-native species have become established in US waters since 1970. Many of these likely became established prior to 1993 when BWE was first introduced. The rate of invasion may have slowed, but Grosholz (2005) estimates there are over 500 different species of non-native organisms now in US coastal waters and furthermore, new introductions are occurring at an increasing rate due to increased shipping and ballast discharge.

To estimate the relevant ballast water discharges to NMFS resources, NMFS analyzed the total coastal and estuarine ballast water discharges in the US over the past six years (excluded the Mississippi River Basin and Great Lakes ports). Ballast water discharge into US waters has increased between 2005 and 2010 with corrected volumes of $123.6x10^6$, $155.5x10^6$, $173.0x10^6$, $192.0x10^6$, $215.1x10^6$, and $236.5x10^6$ tonnes discharged annually from around the globe, respectively. Again, if NMFS adopts the USCG's PVA from the final PEIS, we would expect a 37% reduction in the number of new non-native species under the numerical standard and an 85% reduction under the ballast management program's published goal. Therefore, using the six year average, an invasion rate of approximately 9.7 non-native species per year ($8.47x10^{-8}$ x $182.6x10^6$ x 0.63) would be expected in US waters every year. And as we did for local port case studies, observing the increasing trend in discharge volumes, there is an increase of approximately 22,582,000 tonnes per year each year beyond 2010, increasing the risk of an invasion by approximately 1.91 new non-native species each year. In other words, if the rate of 9.7 new species per year is correct, given the upward trend in ballast discharge provided by the National Ballast Information Clearinghouse, the potential for an invasive species to establish in 2012 would be approximately 13.52 new non-native species each year. On the other hand, if the USCG ballast water management program increases the standard to 1000 times more stringent than the proposed numerical standard, the anticipated rate of invasions would fall to about 3.005 individuals per year nationwide.

Response Analysis

Consequences of exposing listed species and designated critical habitat

The preceding section of this Opinion presented the evidence that leads us to conclude that endangered or threatened species and proposed or designated critical habitat under NMFS jurisdiction are likely to co-occur with discharges of ballast water on, over, or near U. S. waters. In this section, we summarize information on the probable physical, physiological, behavioral, social, and ecological responses of endangered or threatened species or constituent elements of critical habitat given exposure to potentially invasive organisms. Our purpose is not to provide a comprehensive review of the probable responses of proposed, endangered, or threatened species to these stressors; instead, our intention is to identify the range of representative responses we would expect proposed or listed species and their critical habitats to exhibit given exposure to these stressors. We do provide probability estimates of harmful invasions at the end of the Integration and Synthesis section.

Direct effects to listed species

Invasive species are a major threat to many ESA-listed species. For species listed by the USFWS, 26% were listed partially because of the impacts of invasive species and 7% were listed because invasive species were the major cause of listing (Anttila *et al.* 1998). Pimentel *et al.* (2004) found that roughly 40% of US listed species are at risk of becoming endangered or extinct completely or in part due to invasive species, while Wilcove *et al.* (1998) found this to be 49%, with 27% of invertebrates, 37% of reptiles, 53% of fishes, and 57% of plants imperiled partly or wholly due to non-native invasions. In some regions of the world, up to 80% of species facing extinction are threatened by invasive species (Yan *et al.* 2002, Pimentel *et al.* 2004). Clavero and Garcia-Bertro (2005) found that invasive species were a contributing cause to over half of the extinct species in the IUCN database; invasive species were the only cited cause in 20% of those cases.

Introduction of non-native species was found to be a leading cause of endangerment in some western US areas (Flather *et al.* 1998). Richter *et al.* (1997) also identified invasive species as one of three top threats to threatened and endangered freshwater species in the US as a whole, but particularly so in the western US, where recovery of 60% of listed fishes is impaired due to non-native species.

In Hawaii, 44 species of native freshwater fishes are endangered partly due to the 33 non-native species introduced there, while another 27 native species have been negatively affected by these introductions(Maciolek 1984, Pimentel *et al.* 2004). Once invasive species have established, they can potentially migrate widely into surrounding regions. Mitten crabs have been documented to occur up to 1,000 km upstream of their likely establishment origin and are currently an invasive species in the Chesapeake and San Francisco Bays (Leichsenring and Lawrence 2011). Similarly, the Asian kelp expanded its range into San Francisco Bay following its initial establishment in southern California and Mexico (Whitman Miller *et al.* 2004).

Pathogens, Parasites, and Toxins

The impacts of introduced pathogens in the aquatic environment has been poorly explored and we likely know very little about the true frequency and significance of pathogen invasions (Drake *et al.* 2001). Although most available literature addresses pathogens as an invader, Dobbs and Rogerson (2005) contend that free-living microbial organisms lack a true range due to their potentially high reproductive rates, broad tolerances to physical conditions, and ability to form resting stages and, as such, can be considered cosmopolitan and not invaders. A similar cosmopolitan nature has been suggested for bryozoans, barnacles, polychaete worms, hydroids, and molluscs that reside on marine debris (Barnes 2002). Despite these hypotheses, NMFS would expect that if this assertion were true, the occurrence of novel diseases would not occur on a geographic basis or, in the present situation, in association with relatively high levels of ballast water discharge, which a majority of the available literature contradicts. Furthermore, although a pathogen may already exist in waters receiving ballast water discharges, some pathogens such as viruses and bacteria occur in different strains which may be novel to potential hosts, thereby exacerbating the potential for morbidity and/or mortality from a "native" pathogen (Cohen 2010).

Diseases. There are several pathogens that have adverse effects to native fish populations. Viral hemorrhagic septicemia virus was initially introduced to North America from western Europe via ballast water discharge in the Great Lakes. Following its introduction, it caused extensive morbidity and mortality to both farmed and wild fish populations including salmonids, smelts, and rockfish, but not sturgeon (McKenna 2007, Bain *et al.* 2010). Since its establishment during or before 2005, the virus has been found widely in the Northern Hemisphere.

Pathogens also have adverse effects to invertebrate communities. Molluscs such as black and white abalone seem to be particularly sensitive to pathogens. Various species of the genus *Vibrio*, known to cause cholera in humans, white pox and white plague type II diseases in corals, and mortality in abalone of the same genus as black and white abalone, have been identified in ports and ballast water of vessels (Anguiano-Beltrán *et al.* 1998, Ben-Haim and Rosenberg 2002, Aguirremacedo *et al.* 2008). Oyster species have sustained several outbreaks from invasive pathogens, including *Haplosporidium nelsoni* (the cause of MSX disease, which Chesapeake Bay eastern oysters have shown 75-92% mortality to) and *Perkinsus marinus* (the cause of Dermo disease) in California, eastern North America, and Europe (Ford and Haskin 1982, Andrews 1984, Burreson *et al.* 2000, Renault *et al.* 2000, Burreson and Ford 2004), *Bonamia ostreae* in Europe (Van Banning 1987, Ciguarria and Elston 1997), and in the northeastern US, respectively (Ford 1996).

As described in the *Status of Listed Resources*, elkhorn and staghorn coral face severe declines in part due to morbidity and mortality stemming from disease. The introduction of novel pathogens into a habitat can cause overwhelming infections, particularly when coupled with stresses of a disturbed environment which corals must already cope with (Lesser *et al.* 2007).

Parasites. Parasites can directly affect fish species, including listed fish. The Asian parasitic copepod, *Neoergasilus japonicas*, began parasitizing endangered cichlid species in Mexico upon its introduction there (Suárez-Morales *et al.* 2010). When salmon fry were imported to Sweden in the 1950's, a parasitic fluke worm accompanied the fry and infected native Norwegian wild salmon stocks, resulting in 95% mortality among the naïve natives (Josefsson and Andersson 2011).

Like fish, invertebrates are highly susceptible to introduced parasites and there are a number or examples of adverse effects. A parasitic worm, *Terebrasabella heterouncinata*, attacked red abalone in California aquaculture subsequent to the parasites' introduction, significantly reducing growth rate. The pathogen escaped into the marine environment from waste water and shells discharged from these facilities (Cohen and Webb 2002, Culver and Kuris 2002, Bower 2006) and although ballast water was not the transfer vector in this case, it is possible for vessels to transport this pathogen within California waters or re-introduce it to other parts of the US. The aquaculture impacts likely highlight the potential for morbidity and mortality in the wild because *T. heterouncinata* is likely capable of infecting most or all California abalone species, including black and white abalone (Kuris & Culver 1999). A significant risk to black abalone's survival and recovery is a disease called withering syndrome, caused by a Rickettsia-like bacterium, *Xenohaliotis californiensis* that may have been introduced from Asia to southern California via ballast water discharge (Friedman *et al.* 2000)(Smith *et al.* 2003, Bower 2009, Cohen 2010).

Competitors

There are several good examples of introduced species competing with listed species. In the case of listed fish, Waknitz *et al.* (2003) examined what impacts introducing Atlantic salmon along the US west coast would have on Pacific salmon species, many of whom are listed under the ESA. Atlantic salmon escapees from aquaculture could potentially dilute or alter the gene pool of Pacific salmon species, directly compete with native salmon, introduce new diseases to naïve populations, and compete for limited resources. The introduction of brook trout into freshwater systems of the western US, particularly the Columbia River Basin, has directly resulted in the displacement of native trout in these locations through competitive interaction and hybridization (Dunham *et al.* 2002, Feyrer *et al.* 2003). Mills *et al.* (2004) used laboratory experiments to show competitive interactions between least chub and mosquitofish led to reduced survival of the native least chub. Other studies have investigated the impacts of mosquito fish with native cyprinids (Alcaraz *et al.* 2008).

Like with fish, there are examples of invertebrates being introduced that directly compete with native invertebrates, and sometimes listed species. Stat and Gates (2008) found a non-endemic symbiotic dinoflagellate of the same genus used by elkhorn and staghorn coral were present in a Hawaiian Acroporid coral; genetic analyses suggest the symbiotic dinoflagellate was transported to and established in the isolated Northwestern Hawaiian Islands National Monument via ballast water.

Prey

Ballast water potentially introduces species that become prey for native biota in the region. For instance, Hawaiian green sea turtles historically fed upon native algal species, but have recently included the non-native algal species *Acanthophora spicifera*, *Hypnea musciformis*, and *Gracilaria salicornia* in their diets. The contribution of these three alga to green sea turtle diets in the area are now so significant that they compose 40% of the diet among individuals studied (Russell and Balazs 2009). Another two invasive algal species comprised an additional 18% of the diet (Russell and Balazs 2009).

Predators

Ballast water discharges sometimes contain invasive species that are generalist predators. The comb jellyfish, *Mnemiopsis leidyi*, was introduced into the Black Sea in the late 1980's as a result of ballast water discharge and was estimated to constitute 100 million tons of biomass in the Black Sea in 1994. This rapid increase in population caused a decline of native species due to predation (mostly copepods and cladocerans), the extinction of some native pontellid copepods, declines in benthic zooplankton, 70% declines in inshore fish larvae (also prey to the invasive species), benthic anoxia, mucus within the water column, and reductions in biodiversity as a whole (Vinogradov *et al.* 1989, Tsikhon-Lukanina and Reznichenko 1991, Zeitsev and Ozturk 2001, Moncheva and Kamburska 2002). This species preyed on native larval fish, while also effecting several areas within the native food web. If a species having similar ecological impacts was introduced to the US, there is the chance larval stages of listed fish could be consumed by an invasive predator.

Direct effects to designated critical habitat and indirect effects to listed species (ecological responses)

In general, species located higher within a food web (including most ESA-listed species under NMFS' jurisdiction) are more likely to become extinct as a result of an invasion; conversely, species that are more centrally or bottom-oriented within a food web are more likely to establish (Harvey and May 1997, Byrnes *et al.* 2007). Propagule pressure is generally the reason for this trend, as individuals lower in the food web tend to have higher fecundity and lower survival rates (r-selection). This unbalancing of food webs makes subsequent introductions more likely as resource utilization shifts, increasing resource availability, and exploitation success by non-native species (Barko and Smart 1981, Byrnes *et al.* 2007). Such shifts in the base of food webs fundamentally alters predator-prey dynamics up and across food chains (Moncheva and Kamburska 2002). The number of extinction events seems to be roughly correlated with the number of invasive establishments within an area (Harvey and May 1997).

Marins *et al.* (2010) found that port environments, compared to natural areas, had greater species diversity owing to the presence of invasive species. Many invasive species, such as Chinese mitten crabs, zebra mussels, the ascidian *Pyura praeputialis*, and the polychaete *Ficopomatus enigmaticus*, act as ecosystem engineers that become instrumental in the environments in which they invade, changing metabolic and trophic interactions of predator-prey relationships as well as altering physical and chemical processes within an environment (Colwell 1996, Castilla *et al.* 2004, Hecky *et al.* 2004, Wallentinus and Nyberg 2007). However, while a large majority of available information indicates that invasive species alter the ecosystems into which they invade, this is not always the apparent result (Hewitt and Inglis 2004). For instance, changes in native species abundance appears to facilitate the expansion of other invasive species (Daehler and Strong 1996). Grosholz and Ruiz (1996) studied the impacts of green crabs at three different sites, finding that ecological impacts were similar at all three sites, although habitat use and expansion rate exhibited considerable variation.

Pathogens, Parasites, and Toxins

Primarily, pathogens and species with toxic effects have direct effects to listed species, however there are instances when these species affect PCEs of listed species critical habitat or indirectly affect the species in other ways. There are a number of non-native species that have the potential to either expel toxins at low levels, only becoming problematic for other members of the ecosystem if their population grows to very large sizes, resulting in very large amounts of toxins being released. In other cases, pathogens are introduced to an environment affecting organisms in the environment that would directly affect critical habitat PCEs or indirectly affect listed species. Pathogens are in some cases very specific to hosts, but when a species similar to a listed species is introduced, eventually that parasite that specific to the non-native species can shift to also affecting similar native populations. In these cases, the effects may be directly adverse to listed species or indirect to food resources as identified in a species' critical habitat. And in other cases, parasites can have direct effects to PCEs of designated critical habitat or indirectly affect listed species.

Toxins. Red tide dinoflagellates have been introduced via ballast water discharges and have the potential to undergo extreme seasonal population fluctuations. During bloom conditions, high levels of neurotoxins are released into local and regional surface water and air that can cause illness and death in fishes, sea turtles, marine mammals, and invertebrates (as well as their larvae) (Hallegraeff and Bolch 1992, McMinn *et al.* 1997, Hallegraeff 1998, Hamer *et al.* 2000, Hamer *et al.* 2001, Lilly *et al.* 2002).

The dinoflagellate, *Alexandrium catenella*, was likely introduced to Puget Sound via ballast water discharge and is now a major source of paralytic shellfish poisoning and harmful algal blooms in the region (Moore *et al.* 2011). Another species potentially introduced by ballast water, *Pseudonitzschia australis*, resulted in bird mortality in Monterey Bay (Fritz *et al.* 1992) and *Thalassiosira* spp. have been introduced via ballast water, damaging the gills of fishes (including Atlantic salmon) and mollusks when they bloom (Takano 1956, Kent *et al.* 1995, Miyahara *et al.* 1996, Macleod 2002). In the Black Sea, invasive dinoflagellates such as *Gymnodinium uberrimum* and *Alexandrium monilatum* have become dominant phytoplankton, peaking at up to 70% of the biomass compared to native dinoflagellates, clogging fish gills and increasing turbidity (Moncheva *et al.* 1995)(Moncheva and Kamburska 2002). The brown alga, *Aureococcus anophagefferens*, causes brown tide when it blooms and has been found to be transported via vessels along the US east coast, causing diebacks of eelgrass habitat due to blooms decreasing light availability and failure of scallops and mussels to recruit (Doblin *et al.* 2004).

Diseases. Parasites and diseases in some cases are not able to invade without a specific host also invading. However, once established with their invasive hosts, they can infect native, naïve hosts. This was demonstrated along the US Pacific Coast where abalone of the same genera as listed white and black abalone resulting in 50% mortality or higher (Chew 1990, Ciguarria and Elston 1997)(Simon *et al.* 2006)(Nell 2002, Lleonart *et al.* 2003, Nehring 2006b, Haupt *et al.* 2010). In another case, an invasive oyster was blamed as a vector for carrying the protozoan, *Bonamia ostreae*, which infected native oysters and resulted in 50-80% mortality in those infected, severely reducing the abundance of native oysters (Chew 1990, Ciguarria and Elston 1997, Minchin and Rosenthal 2002).

The introduction of American signal crayfish into Swedish lakes resulted in an 80% reduction in the native noble crayfish's abundance, due primarily to the crayfish plague (*Aphanomyces astaci*) that accompanied the invasive crustacean (Minchin and Rosenthal 2002, Josefsson and Andersson 2011). In Britain, the inadvertent introduction of the same species may have been the vector carrying a fungal disease that caused the extirpation of native crayfish(Minchin and Rosenthal 2002).

Parasites. Parasites have the ability to reduce healthy populations of species that may benefit listed species or their critical habitat in some way. A parasite of eels in Japan recently appeared in eels along South Africa, likely as an initial result of ballast water discharge (Christison and Baker 2007). A commonly-occurring nematode parasite of Asian eels infected and caused high mortality in the native eel *Anguilla anguilla* when the former was introduced into European rivers (Minchin and Rosenthal 2002).

Predators. There are a few examples of indirect predatory effects caused by invasive species. European green crabs have invaded both the east and west Coasts of the US, having trophic scale effects to both environments. In Massachusetts Bay, green crabs prey upon native mussels and oysters, altering community structure (Lafferty and Kuris 1996, Grosholz 2002, Pimentel *et al.* 2004). Along the US west coast, green crabs can quickly become the dominant predator of native clams, exerting strong top-down control of at least 20 invertebrate species (Grosholz and Ruiz 1995), with some species showing five to ten fold declines in abundance within the first three years of the invader's detection (Grosholz *et al.* 2000, Grosholz 2002). In both cases, the suppression of these native invertebrates led to increases in their natural prey; however, organisms at higher trophic levels did not increase in response to the green crabs.

In the early 1980s, the ctenophore *Mnemiopsis leidyi* invaded the Black Sea via ballast water discharge, where only three other endemic gelatinous zooplankton existed (Kideys *et al.* 2005). By the end of the decade, the invader was in such high abundance that it had become the dominant zooplankton predator, causing abnormal changes in plankton dynamics and subsequent collapses in pelagic fish stocks (Kideys *et al.* 2005).

The amphipod, *Gammarus tigrinus*, introduced via ballast water, interacts with the native opossum shrimp along Northern Ireland through direct predation on the native and also by facilitating predation by fishes, while the shrimp itself preyed upon the amphipod heavily in areas where it had become dense (Bailey *et al.* 2006). Decapods as a whole reduce the presence of macrophytes, snails, and other benthic animals (Strayer 2010). Also, *P. macrodacylus* may act as a new prey for bethivore generalists, such as shortnose sturgeon.

Competitors. The most commonly reported impact of non-native species in the freshwater and coastal environment is competition for limited resources (Nyberg 2007). Molluscs, decapods, and aquatic plants as taxonomic groups tend to be especially capable invaders and have proven to be disruptive to food webs. The most common impacts are alteration of habitat and nutrient availability as well as altering species composition and diversity within an ecosystem (Strayer 2010). Crabs, polychaetes, and mussels can increase bioturbation and aerate the sediment (Nyberg 2007). Gastropods can alter the biogeochemical cycle through excretion of biogenic silicate in the faeces and pseudofaeces (Ragueneau *et al.* 2005). Molluscan invasions can also provide substrate for epibionts, shelter for benthic species, remove nutrients from the water, decrease turbidity and increase light penetration, remove sediments, and promote phytoplankton blooms by releasing nutrients from sediments (Bertness 1984, Gutierrez *et al.* 2003, Hecky *et al.* 2004).

Invertebrates. There are many examples of invertebrate competition either indirectly affecting similar species to listed species under NMFS jurisdiction or directly affecting the habitat they rely on. The compound tunicate, *Botrylloides sandiegensis*, was released near Woods Hole, Massachussetts and has outcompeted other encrusting organism in the coastal environment of southern New England (Lafferty and Kuris 1996). The invasive snail *Batillaria attramentaria* outcompetes a native species, *Cerithidea californica*, along northern California via a more efficient ability to convert food into body tissue and possibly greater dispersal potential (Savino and Kolar 1996, Carlton 1999, Pothaven *et al.* 2001). The slipper limpet (*Crepidula fornicata*) is a voracious feeder, outcompeting native oysters and scallops in Europe, where it is invasive (Wallentinus and Nyberg 2007). The altered nutrient availability that this species produces may contribute to harmful algal blooms (Wallentinus and Nyberg 2007).

Intentional introductions also provide an opportunity to study the impacts of non-native species on native communities. Aquaculture is often responsible for intentional introductions, such as farmed short-necked clams (*Tapes philippinarum*), which can occur in such high densities that anoxic conditions in the benthos and water column (which can cause extensive mortality of many marine organisms) can occur as bacteria bloom in response the large amount of pseudofeces the clams produce (Bartoli *et al.* 2001, Wallentinus and Nyberg 2007). Additionally, in these farmed areas, other macrobenthic organisms also occur in relatively low abundance. Pacific oysters, also intentionally introduced in many locations, can fundamentally alter the nutrient composition and phytoplankton availability in the habitats they occupy and monitoring of their aquaculture introductions has shown they establish under a variety of environmental conditions (Nehring 2006b, Molnar *et al.* 2008).

Filter feeding benthic species such as clams and oysters convert nutrients from the water column and deposit them in the benthos (Nehring 2006b). Pacific oysters are converting mud flats along Germany and the Netherlands into oyster reefs (Nehring 2006b). A mollusk of the genus *Rapanus* has been blamed for the decreased abundance of native oysters in some areas of the Black Sea (Vinogradov *et al.* 1989). The Asian mussel, *Musculista senhousia*, established a mat over mud flats in Mission Bay, San Diego forming a unique habitat that increased species richness. Tanaids, gastropods, amphipods, insect larvae, and polychaetes and those species that prey upon them appear to have most benefitted from this altered habitat (Crooks 1998, Grosholz 2002). However, larger bivalves and eelgrass appear to be inhibited by the mat covering the mud flats (Crooks 1998). In addition to altering habitat and shifting species distributions and abundances, this Asian mussel also caused increased sediment and organic accumulations converting sand flats into mud flats (Crooks 1998).

Maybe the most well known example of a ballast-mediated invasive species and its subsequent environmental impacts is the zebra mussel, *Dreissena polymorpha*, which established in Lake St. Clair from Europe and have spread to most freshwater systems in the eastern US (Benson 1995, Strayer 1999, Pimentel *et al.* 2004). Shortly after zebra mussels were introduced, their close relative the quagga mussel was introduced via ballast water discharge into Lake Michigan and have become very abundant and pose significant impacts to habitat there, even outcompeting zebra mussels in some areas (Ciborowski 2007). The quagga mussel occupies deeper regions of the Great Lakes and has not spread to other freshwater systems in the US.

Zebra and quagga mussels displace native species in freshwater habitat to the point of local and regional extinction. A baseline study of the Detroit River found that 97% of shells collected were from live, native unionid species, but this declined to only 10% a decade later after these invasive dreissenid mussels invaded the river (Schloesser *et al.* 1998). A similar decline from 84% to 3% occurred along nearby Lake St. Clair over the same time frame, virtually extirpating at least eight native mussel species (Schloesser *et al.* 1998). Nalepa *et al.* (1996) had similar, although not as dramatic findings in reduced diversity of native mussel species. Michigan Technological University (2010) reported that zebra mussel numbers are so large that they can remove significant amounts of phytoplankton from the lake that fish once fed upon. Both *Dreissena* species compete with native taxa for space and food.

Zebra mussels compete with native mussels, clams, and even snails for space, reaching densities as high as 700,000 per square meter (Benson 1995, Keniry 1995, Drake and Bossenbroek 2004, Pimentel *et al.* 2004). Overall, habitats into which zebra mussels invade experience a decline in biodiversity (McNickle *et al.* 2001) and an increase of waste products and bacteria (Strayer 1999). Furthermore, when zebra mussels excrete, they are transferring nutrients from being pelagically available in the water column to the benthos and generally unavailable to fish species. Native Great Lake unionid species, after having zebra mussels grow on them, fail to open and close their shells and thus are unable to burrow, have impaired valve movement, deformed valve margins, and depleted prey (Strayer 1999).

Zebra mussels are also highly effective filter feeders and between that and their abundance compete with native species for available nutrition in the water column (Strayer 1999, Martel *et al.* 2001)(Roberts 1990, Ciborowski 2007, Falkner *et al.* 2009). As a consequence, nearly all native unionid mollusks and the amphipod, *Diporeia hoyi*, have disappeared from the four lower Great Lakes, Lake St. Clair, as well as the St. Lawrence River (Strayer 1999). *Diporeia* was a major component of lake whitefish diet, which is a primary component of many predatory salmonid diets including the economically important lake trout. But as *Diporeia* declined, the condition of lake whitefish and sculpin declined (McNickle *et al.* 2001, Pothaven *et al.* 2001) leading many researchers to suggest the four lower Great Lakes may be on the verge of trophic collapse (Nalepa *et al.* 2006, Riley *et al.* 2008, Pothoven *et al.* 2011).

Related to the zebra mussel’s ability to effectively filter feed is increased water clarity, altered benthic and water chemistry, as well as blue-green and benthic algae blooms and increases in gastropod populations (Strayer 1999, Janssen and Jude 2001, McNickle *et al.* 2001). Their feces, because there is so much of it, could lead to cyanobacterial blooms (producing toxins) in sediments, causing waters to become anaerobic leading to mass mortalities of resident fish populations and the upper trophic levels, with other tertiary effects possible within the ecosystem (Vanderploeg *et al.* 2001, Ciborowski 2007). These impacts have generally been observed throughout the Great Lakes region (Schloesser *et al.* 1998).

The Asiatic clam, *Corbicula fluminea*, significantly reduces native phytoplankton abundance and can be found in excess of 10,000 individuals per square meter, which likely reduces the fitness of native filter feeders in the area due to competition for space and food resources, causing severe declines or local extirpations of native mussels in some areas Kennedy and Huekelem 1985, Gardner *et al.* 1976, Ilarri *et al.* 2011, Fuller and Imlay, 1976; Clarke, 1986(Cohen *et al.* 1984, Leff *et al.* 1990, Strayer 1999, Cordeiro *et al.* 2007). In addition to filtering food from the water column, young Asiatic clams also feed on nutrients in the sediment, potentially competing with benthic sphaeriids (Strayer 1999). When *C. fluminea* experiences die-offs, the amount of ammonia produced by decomposition may adversely affect surrounding native taxa Cooper *et al.* (2005a, 2005b). Overall, Asiatic clams can fundamentally alter the ecosystems into which they invade (Ruiz *et al.* 1997).

The invasive green mussel *Perna viridis* may competitively displace the native scorched mussel *Brachidontes exustus* through its greater growth rate and maximum size in Tampa Bay (Ranwell 1964). *Mya arenaria*, an invasive soft-shelled clam, may be the most common large clam in many embayments along the Pacific Northwest coast (Carlton 1999). *Mytilus galtoprovincialis*, an invasive mussel, has displaced the native mussel to become the dominant mussel throughout the Western Cape region of South Africa (Carlton 1999). The invasive Japanese mussel, *Musculista senhousia*, has outcompeted numerous native species of coralline algae, serpulid polychaetes, bryozoans, and hydroids to exclusively occupy large patches of intertidal and subtidal rocky habitat along New Zealand ((Carlton 1999).

Vegetation. Invasive plants can cause widespread habitat alteration, including native plant displacement, changes in benthic and pelagic animal communities, altered sediment deposition, altered sediment characteristics, and shifts in chemical processes such as nutrient cycling (Grout *et al.* 1997, Wigand *et al.* 1997, Ruiz *et al.* 1999). Introduced seaweeds alter habitat by colonizing previously unvegetated areas, while algae form extensive mats that exclude most native taxa, dramatically reducing habitat complexity and the ecosystem services provided by it (Wallentinus and Nyberg 2007). Invasive algae can alter native habitats through a variety of impacts, including trapping sediment, reducing the number of suspended particles that reach the benthos for benthic suspension and deposit feeders, reduce light availability, and adversely impact foraging for a variety of animals (Gribsholt and Kristensen 2002, Britton-Simmons 2004, Levi and Francour 2004, Sanchez *et al.* 2005). The green alga *Caulerpa taxifolia* can become so dominant in Mediterranean areas as to form dense mats and exclude other algal species during some seasons and alter the diversity of other species present, including fishes such as red mullet, which could not forage in certain areas due to physical barriers created by the alga (Boudouresque 2002, Levi and Francour 2004, Wallentinus and Nyberg 2007). The dense aggregations of this alga impair suspended particles reaching the bottom, reducing available food for benthic feeders (Wallentinus and Nyberg 2007). *Sargassum muticum* and *Gracilaria vermiculophylla*, can form vegetative cover in areas that were previously lacking such cover, attracting fishes and other species that enhance local biodiversity (Nyberg 2007).

The complexity of an invaded system can also be altered due to changes in flow regimes and other factors (Wallentinus 2002, 2006b, a). Particularly significant for juvenile salmon is that some of their invertebrate prey are adapted to take advantage of seasonal spikes in available detritus, which invasives like purple loosestrife can disturb (Grout *et al.* 1997). Some invasive or nuisance alga species suppress the growth of native species via release of chemicals (Guerriero *et al.* 1992, Friedlander *et al.* 1996, Hellio *et al.* 2004, Råberg *et al.* 2005, Nyberg 2007, Wallentinus and Nyberg 2007). *C. taxifolia* produces toxic metabolites that negatively influence other plants (Wallentinus and Nyberg 2007). Overall, *C. taxifolia* is viewed as highly impactful to habitats it invades due to its ability to reduce the overall biodiversity of native species in the area (Molnar *et al.* 2008).

A red alga, *Gracilaria salicornia*, that established in Hawaii has led to significant changes in coral reefs there. *G. salicornia* reduced irradiance by 99%, increased sedimentation, and changed diurnal dissolved oxygen and pH cycles on the surfaces it established on. Some species may benefit from this, as it creates cover for several filter-feeding and detritivore organisms, but smothered benthic sessile organisms, notably the most common coral species in these habitats, as they form the basis for the continued existence of coral reefs that all other species in the habitat ultimately rely upon (Martinez 2012). Two other red algae, *Acrothaninion preissii* and *Womersleyella setacea*, form dense mats that can impede sedimentation (Wallentinus and

Nyberg 2007). Another red alga in Hawaii, *Hypnea musciformis,* competes with native coral for space on coral reefs (Lafferty and Kuris 1996).

Many algal species also alter ecosystems by reducing the amount of light available for other plants. The invasive brown alga, *Sargassum muticum,* can shade large areas and reduce available light for native algal taxa (Britton-Simmons 2004, Sanchez *et al.* 2005, Wallentinus and Nyberg 2007). As a result, some kelp species in the San Juan Islands can be outcompeted (reductions of 50-75%) and green sea urchin abundance can decline significantly with the loss of this primary food, potentially destabilizing kelp ecosystems that entire biological communities are dependent upon (Britton-Simmons 2004) (Ambrose & Nelson 1982). Invasive freshwater plants in Sweden have reduced available light, outcompeted native plants, altered water chemistry, and reduced biodiversity where they have invaded (Josefsson and Andersson 2011).

Once intentionally planted in Hawaii, mangroves spread from the island of Molokai throughout the main Hawaiian Islands (Allen 1998). An unintended consequence of the mangrove introduction was that they completely fill and shade the shallow pools they establish in, leading to these pools filling in much more quickly than they otherwise would (up to several centimeters of sediment annually), likely heavily impacting the algal communities that may occur there and altering (either positively or negatively) dissolved oxygen levels. These changes can affect other aspects of the ecosystem including decreased biodiversity, impeding the recovery of endangered waterbirds, and disproportionally providing habitat for invasive species over native taxa (Allen 1998).

Other invasive plant species can be beneficial for native taxa in other ways (Posey 1988)(Bulleri *et al.* 2006. Invasive marine plants can stabilize sediments with their roots, increase the oxidizing capacity of sediments, reduce wave disturbances, improve water quality, and transform beaches into marshes (Posey 1988, Allen 1998, Ronnback 1999, Bruno 2000, Bruno and Kennedy 2000, Gribsholt and Kristensen 2002)(Cuddington and Hastings 2004). The invasive green alga *Codium fragile* promoted recruitment of native mussels (Bulleri *et al.* 2005).

The genus *Spartina* has invaded wetlands throughout the world, causing diverse impacts to its new habitats (Wallentinus and Nyberg 2007). Lower biodiversity has been associated with *Spartina* spp. introductions (Carpenter 1987). In Britain, *S. anglica* has invaded eelgrass habitat, reducing the abundance of eelgrass and thus food availability for waterfowl (Daehler and Strong 1996). In addition, the introduction of a variety of *Spartina* marshgrass to England via ballast water discharge where a similar native species occurred resulted in a hybrid species, which is now the dominant species within its habitat (Thompson 1991). In the Netherlands, along Oregon, and in San Francisco and Humbolt Bays, this species as well as *S. alterniflora, S. densiflora,* and *S. patens* have outcompeted native plants (Daehler and Strong 1996)(Chen *et al.* 2004). Because *Spartina* spp. can occupy habitat that native grasses cannot, it establishes in mudflats that can increase available habitat for benthic invertebrates (Paulo da Cunha Lana 1991, Daehler and Strong 1996).

Spartina spp. and *Zostera japonica* stabilize sediments in higher energy environments (Wallentinus and Nyberg 2007). As a result, *Spartina* spp. frequently transform higher-energy habitats like beaches into lower-energy environments (marshes) (Carpenter 1987, Bruno 2000, Bruno and Kennedy 2000). *Z. japonica* displaces native eelgrass, speeds decomposition rates, and limits nutrient release into the water column along the US west coast (Wallentinus and Nyberg 2007). Since sediment is trapped to a greater degree within *Spartina*-dominated areas than many other grasses or mud flats, tidal flows can be decreased in these areas (Ranwell 1964, 1967, Christie 1974, Carpenter 1987, Daehler and Strong 1996). This shift from high energy environments to low energy environments has obvious implications for ecosystem change, which are not always detrimental to natives. For example, more rooted plants can establish in lower-energy environments (Bruno 2000, Bruno and Kennedy 2000, Wallentinus and Nyberg 2007).

Fish. Trophic cascades can result from the introduction of invasive fishes (Power 1990, 1992, Ruiz *et al.* 1999). Round gobies introduced to the Gulf of Gdansk in the Baltic Sea on or before 1987 via ballast water became the dominant shallow water fish in at least the western portion of the Gulf within a decade and continue to spread through the region, using all available solid substrates (Sapota and Skora 1999, 2005). This may have been aided by the anthropogenicly-induced loss of predators (Sapota and Skora 1999). Round gobies have also established in the Great Lakes, where they compete for prey resources with native species, prey upon native competitor eggs and young directly, and aggressively exclude native fishes from their individual territories (Thomas 1997, French and Jude 2001, Janssen and Jude 2001, Carman *et al.* 2006, Bergstrom *et al.* 2008). Although the round goby has a specialized diet in its native range, it has shown the ability to shift to be a general predator in some invasive populations, including foraging on invasive zebra mussels in the Great Lakes (Thomas 1997, Carman *et al.* 2006, Lederer *et al.* 2006). By outcompeting native species, there is a shift of nutrient availability from the pelagic zone to the littoral zone.

The sea lamprey (*Petromyzon marinus*) was introduced into the Great Lakes and has seriously altered the abundance of lake trout and lake whitefish(Janssen and Jude 2001). The Eurasian ruffe was released into the Great Lakes via ballast water discharge and has become a major predator of native benthic invertebrates and competes for available prey and habitat with native taxa, such as yellow perch and preys upon the eggs of lake herring (Lovell and Stone 2005)(Savino and Kolar 1996, Selgeby 1998, Fullerton *et al.* 2000, Drake 2005). Invasive fishes can compose a large portion of fish taxa in at least some areas, including new Zealand where 53% of fish taxa are exotic, Puerto Rico where invasive fish are 91% of the total species, and Brazil where they are 13% of the total (Lövei 1997).

Habitat Effects. Invasive species have been referred to as one of the top four threats to the world's oceans consistently ranked behind habitat degradation and alteration (Raaymakers and Hilliard 2002, Raaymakers 2003, Terdalkar *et al.* 2005, Wambiji *et al.* 2007) (Pughiuc 2010). In most cases, habitat is directly affected by human alterations, as identified in the baseline section, such as hydromodification, mining, dredging, drilling, and construction. However, invasive species, facilitated by human commerce, have the ability to directly alter ecosystems upon which listed species rely.

The spiny water flea, *Bythotrephes longimanus*, is another example of an invasive species that causes extensive ecosystem disruption (Grout *et al.* 1997). This invader, first introduced by ballast water from northern Europe to Lake Ontario in the early 1980s may eventually occupy thousands of lakes throughout North America (Johannsson *et al.* 1991, Grout *et al.* 1997, Kerfoot *et al.* 2011). *Bythotrephes* is an important contributor to its native habitat, including as prey to salmon; however, in the Great Lakes, they reduce the fitness of many fish that are prey to salmonids (Hessen *et al.* 2011). *Bythotrephes* preys heavily upon plankton species, severely reducing not only their abundance, but has also caused their diversity to decline by roughly 20% (Foster and Sprules 2009, Kerfoot *et al.* 2011)(Rennie *et al.* 2011). As a result, rotifers decline because of reduced diatom food resources and phytoplankton increase because *Bythotrephes* feeds on their competitors (Kerfoot *et al.* 2011) (Hovius *et al.* 2006; Strecker *et al.* 2011). Further tertiary effects include elevation of contaminant levels in higher-level predators due to extensions in the food web that allow for additional contaminants to accumulate in the underlying prey base (Kerfoot *et al.* 2011, Rennie *et al.* 2011). Other macroinvertebrate predators and fishes are also likely adversely impacted by this disruption of their prey base, with less prey available to them (Parker Stetter *et al.* 2005, Foster and Sprules 2009). These alterations to ecosystem food webs appear to be stable and persistent (Yan *et al.* 2008). Through these mechanisms, *Bythotrephes* alone represents a significant threat to the biodiversity within temperate North American aquatic environments (Grout *et al.* 1997).

Dreissena mussels introduced to the Baltic Sea altered sediment composition from soft bottom into shell gravel and even hard substrate due to the abundance of their shells in the benthos (Olenin and Leppäkoski 1999). Furthermore, *Dreissena* and other benthic invaders remove nutrient resources from the water column and transfer them to the sediment in their excrement. In many cases, this decreases the nutrient outflow from some embayments and causes an accumulation of organic material, leading to increased eutrophication and fundamentally altering the ecological function of the habitat. Sometimes increasing the nutrients available in an area works to the advantage of predatory fishes for which additional prey become available (Olenin and Leppäkoski 1999).

An invasive polychaeate, *Ficopomatus enigmaticus*, established burrows in an Argentinean lagoon altering water flow and sedimentation (Wallentinus and Nyberg 2007). However, as above, this habitat modification is not always to the detriment of native taxa. The ascidian, *Pyura praeputialis*, invaded coastal waters along Chile and formed dense matrices in the intertidal zone, providing habitat that was extensively used by native invertebrates and having double the species richness in these habitats than surrounding unaltered habitat (Castilla *et al.* 2004, Wallentinus and Nyberg 2007).

Other invertebrates can also have major impacts on the ecosystems they invade. The invasive bryozoan *Membranipora membranacea* established on kelp fronds in the Gulf of Maine, making them brittle and inflexible and causing them to drop off prematurely (Lambert *et al.* 1992). The invasive polychaete, *Marenzelleria viridis*, was introduced to coastal Baltic Sea lagoons in the mid 1980s as the first polychaete in the ecosystem. Within five years, the abundance of this species resulted in an increase in the biomass present within the benthos by an order of magnitude (Zmudzinski 1996, Zettler 1997, Olenin and Leppäkoski 1999, Leppaekoski and Olenin 2001, Panov *et al.* 2003). This species digs holes in the benthos that are much deeper than the native polycheate (deeper areas), thereby introducing much more oxygen into the benthos and increasing the denitrification process causing changes to the native habitat (Panov *et al.* 2003, Wallentinus and Nyberg 2007). These changes have adversely affected the prey species for native predators.

Asian crabs (*Hemigraspus sanguineus*) compete for space (burrows) with native fiddler crabs along Connecticut (Wallentinus and Nyberg 2007). Mitten crabs form extensive burrows that can enhance mud bank erosion (Rudnick *et al.* 2000, Chinese Mitten Crab Working Group 2003). The Asian copepod *Pseudodiaptomus inopinus* has become the dominant macroinvertebrate in some areas of the Pacific Northwest after establishing there via ballast water discharge (Kipp *et al.* 2010). Oriental shrimp (*Palaemon macrodactylus*) were likely introduced into New York State waters via ballast water and may act as competitors of other species within the ecosystem, although no ecosystem damage has been documented by this invasive species here so far (Warkentine and Rachlin 2010).

Echinoids (sea urchins) cause significant habitat alteration where they invade (Himmelman and Steele 1971, Dean *et al.* 1983, Kitching and Thain 1983, Haupt *et al.* 2010), becoming the dominant benthic grazers. They feed heavily upon kelp beds, diminishing their abundance and the habitat they offer for entire marine communities.

The introduced periwinkle, *Littorina littorea*, ranging along the Atlantic Coast from Canada to the mid-Atlantic, is highly-influential in the sedimentation process; because individuals cumulatively engage in so much grazing, some bottom habitats have become dominated by hard-bottom instead of soft bottom as they formerly were (Bertness 1984, Wallentinus and Nyberg 2007)(Carlton 1999). Significant declines in soft-sediment habitats and fringing salt marshes are attributed at least partially to the invasion of this species, possibly due to consumption of marsh grasses, such as *S. alterniflora* (Bertness 1984). Species normally adapted to living in soft-bottom systems are gradually replaced by species better adapted for hard-bottom substrates.

Case Studies of Invasive Species Impacts to Habitat

San Francisco Bay

As a result of the large numbers of introductions into San Francisco Bay, much of the biologically-available energy that was available to native taxa is now being exploited by invasive species (Cohen and Carleton 1995). With at least 234 established non-native populations in San Francisco Bay, there are many well studied adverse effects to the area. However, the vast majority of the invasive species appear to have benign effects on fish species, such as salmonids, and habitat generally.

Direct Effects to Listed Species. The invasive Atlantic ribbed marsh mussel *Arcuatula demissa* has contributed significantly to the decline of local clapper rails by trapping the bird’s beak in its shell if the rail inserts it into the open shell; rails cannot pull their beaks out and die of starvation (Cohen and Carleton 1995). Adult rails also lose toes by the mussel closing on them as the rails walk through the marsh, although the result for young rails is likely mortality (Moffitt 1941, Josselyn 1983, Takekawa 1993)(Cohen and Carleton 1995). However, the invasive mussel also appears to be prey of the rail to a significant extent (Moffitt 1941)(Cohen and Carleton 1995).

Introduced fishes have also greatly affected the San Francisco Bay ecosystem. Striped bass, largemouth bass, smallmouth bass, bluegill, and green sunfish are all introduced species to the area, although largely through means other than ballast water discharge (Cohen and Carleton

1995). Cohen and Carleton (1995) documented that these fishes have led to the extinction of four native fish species not only in the Bay, but throughout their range either directly through predation or indirectly through competition for prey and/or breeding sites. These predatory fish also impact listed salmonids. And ironically, striped bass themselves are impacted by invasive species, with juvenile abundance declining in association with declines in their primary prey species, mysid shrimp, likely due to effects caused by other introduced species in the Bay (Nobriga and Feyrer 2008).

American bullfrogs, along with some of the fish species above, have predated heavily upon native red-legged and yellow-legged frogs in the Bay and Delta region (Cohen and Carleton 1995). Introduced silverside minnows have also caused the decline in delta smelt via predation on the smelt's eggs and larvae (Cohen and Carleton 1995).

Indirect Effects to Listed Species. The Asian and soft shell clams, *Potamocorbula amurensis* and *Mya arenaria*, respectively, are now highly influential within the bay system, occurring in such high abundance so as to transfer the phytoplankton biomass in the water column to the benthos that is then exploited by benthic invertebrates and bottom-feeding fishes, instead of pelagic species (Alpine and Cloern 1992, Cohen and Carleton 1995, Grosholz 2002). *Mya* spp. alone is estimated to be capable of filtering all phytoplankton in the bay daily (Cohen and Carleton 1995). Like the Asian and soft shell clam, in the southern portion of San Francisco Bay, introduced clams of the genera *Gemma*, *Venerupis*, and *Musculista* control phytoplankton growth with their filtering capacities (Cohen and Carleton 1995).

Overbite clam (*P. amurensis*) abundance is high enough to reduce the availability of plankton in the Sacramento-San Joaquin River Delta (Cohen and Carleton 1995). The Sacramento–San Joaquin River Delta is estimated to have lost 43% of its annual primary productivity between 1975 and 1995, partly as a result of this invasive species (Jassby *et al.* 2002, Feyrer *et al.* 2003, Falkner *et al.* 2009), leading to the decline of native fish species. Alpine and Cloern (1992) provided an even larger estimated decline from 106 g C/m^2 to 39 g C/m^2.

The introduced snail, *Batillaria attramentaria*, is slowly out-competing its native counterpart, *Cerithidea californica*. The invasive Atlantic mudsnail *Ilyanassa* spp. has displaced *Certhidea* snails from mudflats to saltmarsh habitats, with an associated decline in the native species' abundance (Cohen and Carleton 1995). Several species including the Asiatic clam, polyzoan, and hydroid (*Corbicula*, *Urnatella*, and *Cordylophora* spp., respectively) have led to increased sedimentation in canals, reducing flow rates and probably flushing within estuaries (Cohen and Carleton 1995). Native crayfish have been displaced by introduced varieties (Cohen and Carleton 1995). While these are all examples of shifts in species abundance, it is unclear if these shifts have any indirect effect to listed fish in San Francisco Bay.

Direct Effects to Habitat. The introduction of invasive species almost always has an unintended impact to the ecosystem. The direct and indirect effects to listed species described above impact the ecosystem to some extent. Additionally, there are species that may not impact other species in the area, but they may alter habitat in such a way that the introduction make some species less fit. The Australian-New Zealand boring isopod *Sphaeroma quoyanum* creates small holes in mud bank habitat. Since its introduction, this small animal has altered mud banks throughout San Francisco Bay with these bore holes. *Sphaeroma*'s holes are now considered the primary mechanism of bank erosion, causing banks to recede by several meters in some locations since its introduction over a century ago (Cohen and Carleton 1995).

Several invasive plants also impact San Francisco Bay's ecosystem. Peppergrass (*Lepidium latifolium*) may act to displace rare native marsh plants, such as the mudflat quillplant (*Lillaeopsis masoni*) (Cohen and Carleton 1995). The Atlantic cordgrass, *Spartina alterniflora* is invasive throughout many regions of the world and, in San Francisco Bay, may benefit some species like clapper rails, which occur in their highest densities in native cordgrass habitat. Expansion of cordgrass habitat may provide more protective habitat from predators, although competition between the native and invasive cordgrass species may well lead to the decline of the native cordgrass (Cohen and Carleton 1995). Some properties of Atlantic cordgrass lend themselves to providing less habitat for native wetland animals and infauna (Cohen and Carleton 1995). The invasive *Spartina* may also alter sedimentation rates, reduce light levels by its increased canopy density, and loss of shorebird feeding habitat through colonization of mudflats (Cohen and Carleton 1995). *Spartina alteriflora* has hybridized with the native *S. foliosa* in the Bay (Daehler and Strong 1996, Anttila *et al.* 1998).

Chesapeake Bay

Like San Francisco Bay, there are many introduced species in the Chesapeake Bay. A comprehensive review of the impacts of invasive species to the Chesapeake Bay was conducted by Ruiz *et al.* (1999). With at least 196 established non-native populations in the Chesapeake Bay, it is surprising that most of the impacts of invasive species on the Chesapeake Bay are generally undocumented. The authors found that 20% of the 196 documented invasive species had significant ecological impacts, while most of the other invasive species had not been studied for their impacts. Of the 39 species with significant ecological impacts, 69% did so through competition with native species, 38% altered habitat, 44% served as prey, 15% were predators of native species, 21% engaged in extensive herbivory, 8% produced hybrids with native taxa, and 8% were parasitic (Ruiz *et al.* 1999). Plants and fish were the largest taxonomic groups represented in the known invasive species of the Chesapeake Bay, representing 23% and 18% of the invasive species by taxa, respectively.

Direct Effects to Listed Species. There is limited information on direct effects to listed species. The only listed species under NMFS jurisdiction in the Chesapeake Bay are sturgeon and sea turtles. Juvenile sturgeon could be prey to introduced piscivorous fish. There are several introduced fish species (including smallmouth bass, and sunfishes) blamed for declines in native taxa (Ruiz *et al.* 1999) and while undocumented, may impact sturgeon.

Indirect Effects to Listed Species. In the case of the Chesapeake Bay, while the invasive species have not been well studied, it appears the best documentation of effects may be indirect to sturgeon or sea turtles via alteration of food web dynamics and food availability. Two protistan pathogens, *Haplosporidium nelsoni* and *Perkinsus marinus*, are significant contributors to a 90% reduction in oyster abundance in the Chesapeake Bay over the past century, causing secondary effects such as reduced oyster reef habitat and altered food webs (Ruiz *et al.* 1999). The rapa whelk is now an abundant predator of native clams and oysters in the Bay (Deacutis and Ribb 2002) with similar ecological impacts to the protest pathogens. Mud crabs have also declined as a result of the invasive parasitic barnacles, *Loxothylacus panopaei*, which causes reproductive failure in the host (Van Engel *et al.* 1966, Hines *et al.* 1997, Ruiz *et al.* 1999).

The Asiatic clam (*Corbicula fluminea*) is so abundant in the Potomac River that it is estimated this species alone can filter the total phytoplankton biomass in three to four days and can constitute 90% or more of the bivalve biomass in some Bay areas. Such efficient conversion of energy from the pelagic to the benthic environment likely benefits shortnose sturgeon by increasing worms and chironomids, two of their prey items. As a result of this invasion, between 1981 and 1993, water clarity tripled, subsequently increasing aquatic vegetation 50%, and ultimately increasing abundance of fish populations, slowing currents, increasing sedimentation, as well as altering benthic community composition and sediment characteristics through its large production of pseudofeces (Cohen *et al.* 1984, Phelps 1994, Ruiz *et al.* 1999).

Direct Effects to Habitat. Direct and indirect effects to listed species also generally have impacts to habitat. For instance, the description of indirect effects caused by the Asiatic clam also have impacts to habitat in the Chesapeake Bay. The reed, *Phragmites auatralis*, also outcompetes local plants and has become widespread and dominant within the Chesapeake Bay, altering habitat parameters and animal abundances (Marks *et al.* 1994, Ruiz *et al.* 1999). *Typha angustifolia* has similar impacts, outcompeting local species, reducing flow rates, increasing sedimentation, and altering sediment chemistry (Ruiz *et al.* 1999).

Two invasive aquatic plants, *Hydrilla verticillata* and *Myriophyllum spicatum*, have received significant attention in the Chesapeake Bay. They form dense mats, alter aquatic chemical and habitat characteristics, fish and invertebrate communities, compete with native plants, and change the food base available for local waterfowl and fishes (Ruiz *et al.* 1999). Also noteworthy is that the cover provided by *Hydrilla* spp. provides additional refuge for smaller fishes, which can increase the populations of larger predatory species (Killgore *et al.* 1989, Ruiz *et al.* 1999). *Trapa natans,* a floating plant, at one time also outcompeted native plant species to the detriment of fishes and waterfowl, but has not recovered from an eradication program in the 1930s (Ruiz *et al.* 1999).

As has been documented in other regions, there appears to be a correlation between habitat disturbance and the susceptibility to invasions. Jewett *et al.* (2005) experimentally found that low dissolved oxygen levels (common in the Chesapeake Bay) resulted in invasive species establishing in a new area, to the detriment of dominant native taxa prior to low dissolved oxygen levels. Many researchers have also documented the eutrophic conditions of the Chesapeake Bay influence the propensity of species to invade. This situation tends to lead to low light levels, which make some invasive species, such as the submerged and emergent aquatic plants outlined above more competitive with native varieties (Barko and Smart 1981, Grace and Harrison 1986, Marks *et al.* 1994, Ruiz *et al.* 1999).

Puget Sound

Although we searched for information addressing the impacts of invasive species on Puget Sound, we did not find any significant information documenting biological, physical, chemical, or ecological responses of species or habitats in this area to invasive species.

Integration and Synthesis

Consequences of Specific Proposed Actions under the Program

Direct effects to listed species. Little to no information is available supporting direct effects that ballast-mediated invasive species have on species listed under NMFS' jurisdiction. The only direct connection we found supported a beneficial connection of invasive algae providing a significant new food source for Hawaiian green sea turtles (Russell and Balazs 2009). By assessing the adverse effects of disease, predation, and competition to similar species or similar functional groups, we were able to conclude the likely affects of new ballast-mediated invasive species to NMFS trust resources.

It is reasonable to expect that ballast-mediated invasive diseases or parasites can have significant and catastrophic effects, such as that which has occurred to Atlantic salmon in Norway as a result of an invasive fluke being introduced, white pox disease in Acroporid corals, withering syndrome in California abalone, and harmful algal blooms that poison resident marine mammals, sea turtles, and fishes (Hallegraeff and Bolch 1992, McMinn *et al.* 1997, Anguiano-Beltrán *et al.* 1998, Hallegraeff 1998, Friedman *et al.* 2000, Hamer *et al.* 2000, Hamer *et al.* 2001, Ben-Haim and Rosenberg 2002, Lilly *et al.* 2002, Smith *et al.* 2003, Aguirremacedo *et al.* 2008, Bower 2009, Cohen 2010)(Hallegraeff *et al.* 1995)(Josefsson and Andersson 2011). When ballast-mediated invasive species establish in an area occupied in the same time and place as listed species and the result is a disease or parasite, we expect a range of effects from beneficial to harm of an unpredictable number of listed individuals.

Predation on early life stages (particularly egg or larval stages) of listed salmonids, sturgeon, rockfish, eulachon, corals, and abalone is by numerous examples of zooplankton-feeding invasive species such as mussels, silverside minnows, and numerous piscivorous fish has been documented and is a likely consequence of future invasions. When ballast-mediated invasive species establish in an area occupied in the same time and place as listed species and the result is predation, we expect a range of effects from beneficial to harm of an unpredictable number of listed individuals.

Competition is another means by which invasive species are likely to directly affect listed species. Particularly susceptible to direct effects of competition are the listed invertebrates (abalone and corals) because filter feeding and photosynthetic organisms are more likely to become established than larger predatory fish species. Invasive filter feeders along the California coast would compete with abalone for both energy and space, while an invasive autotroph like an alga could grow over coral, out-competing corals for sunlight. While less likely, benthivores and piscivores have established in the past and that is likely to happen again in the future. The response to such ballast-mediated invasions would be competition with sturgeon, sawfish, or salmonid species. When ballast-mediated invasive species establish in an area occupied in the same time and place as listed species and the result is competition, we expect a range of effects from beneficial to harm of an unpredictable number of listed individuals.

Effects to designated critical habitat. Our critical habitat analysis determines whether the consequences of the proposed action will destroy or adversely modify critical habitat for ESA-listed species by examining any change in the conservation value of the essential features of critical habitat. This analysis does not rely on the regulatory definition of "adverse modification or destruction" of critical habitat. Instead, this analysis focuses on statutory provisions of the ESA, including those in Section 3 that define "critical habitat" and "conservation," those in Section 4 that describe the designation process, and those in Section 7 setting forth the substantive protections and procedural aspects of consultation.

The *Response analysis* documents indirect effects to listed species via habitat and ecosystem alteration as the major mechanism by which ballast-mediated invasive species can adversely affect listed species. While this is not the result of every invasion, it is a probable outcome over time. In many cases, this also constitutes adverse affects to critical habitat (specifically of eulachon, salmonids, corals, abalone, Cook Inlet beluga and southern resident killer whales, green and hawksbill sea turtles, and sturgeon) through impairment of PCEs associated with quantity and quality of available prey or suitability of habitat for growth.

The PCEs for each listed species, where they have been designated, are described in the *Status of Listed Resources* section of this Opinion. The PCEs identify those physical or biological features that are essential to the conservation of the species that may require special management considerations or protections. The species addressed in this Opinion have similar life history characteristics and therefore, many of the same PCEs.

The PCEs for gulf and green sturgeon focus on life history stages, rearing, migration, and spawning and contain physical and biological features essential to the conservation of the species or DPS. The PCEs for gulf and green sturgeon are:

1. Abundant food resources for all larval, juvenile, sub-adult, and adult life stages;
2. Water flow necessary for normal behavior, growth, and survival of all life stages;
3. Water quality, including temperature, salinity, oxygen content, and other chemical characteristics, necessary for normal behavior, growth, and viability of all life stages;
4. Amigratory pathway necessary for the safe and timely passage of Southern DPS fish within riverine habitats and between riverine and estuarine habitats;
5. Water depth in holding pools over five meters deep for both upstream and downstream holding of adult or sub-adult fish, with adequate water quality and flow to maintain the physiological needs of the holding adult or sub-adult fish; and
6. Sediment quality (*i.e.*, chemical characteristics) necessary for normal behavior, growth, and viability of all life stages.

These PCEs for salmonids and Pacific eulachon include sites essential to support one or more life stages (sites for spawning, rearing, migration and foraging) and contain physical or biological features essential to the conservation of the ESU/DPS, such as:

1. Freshwater spawning sites with water quantity and quality conditions and substrate supporting spawning, incubation and larval development;

2. Freshwater rearing sites with water quantity and floodplain connectivity to form and maintain physical habitat conditions and support juvenile growth and mobility; water quality and forage supporting juvenile development; and 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;
3. Freshwater migration corridors free of obstruction, along with water quantity and quality conditions and natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, side channels, and undercut banks supporting juvenile and adult mobility and survival;
4. Estuarine areas free of obstruction, along with water quality, water quantity, and salinity conditions supporting juvenile and adult physiological transitions between fresh and saltwater; natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels; and juvenile and adult forage, including aquatic invertebrates and fishes, supporting growth and maturation;
5. 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; and
6. Offshore marine areas with water quality conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation.

These PCEs for southern resident killer whales include sites essential to support one or more life stages and contain physical or biological features essential to the conservation of the DPS, such as:

1. Water quality to support growth and development;
2. Prey species of sufficient quantity, quality, and availability to support individual growth, reproduction, and development, as well as overall population growth; and
3. Passage conditions to allow for migration, resting, and foraging.

The PCE for leatherback sea turtles essential to support population fitness and the conservation of the DPS, is the occurrence of prey species, primarily scyphomedusae of the order Semaeostomeae (Chrysaora, Aurelia, Phacellophora, and Cyanea), of sufficient condition, distribution, diversity, abundance and density necessary to support individual as well as population growth, reproduction, and development of leatherbacks.

These PCEs for Cook Inlet beluga whales are essential to their conservation:

1. Intertidal and subtidal waters of Cook Inlet with depths <30 feet MLLW and within 5 miles of high and medium flow anadromous fish streams;
2. Primary prey species consisting of four species of Pacific salmon (Chinook, sockeye, chum, and coho), Pacific eulachon, Pacific cod, walleye pollock, saffron cod, and yellowfin sole;
3. Waters free of toxins or other agents of a type and amount harmful to Cook Inlet beluga whales;
4. Unrestricted passage within or between the critical habitat areas; and

5. Waters with in-water noise below levels resulting in the abandonment of critical habitat areas by Cook Inlet beluga whales.

These PCEs for Hawaiian monk seal are essential to ensuring the health, well-being, and continued viability of their populations:

1. Pupping and major hauling beaches including the vegetation backing the beaches;
2. Shallow protected water adjacent to the above;
3. Deeper inner reef areas and lagoon waters;
4. Other waters surrounding the northwest Hawaiian Islands to at least 80 fathoms; and
5. Banks and shoals without emergent lands and pelagic waters.

We can draw several generalizations from the *Response analysis* that summarize the effects that ballast-mediated invasive species have on their new habitat. Benthic invertebrates, such as mussels, polychaetes, and hydroids become dominant filter feeders, greatly reducing the amount of organic energy that is available to native taxa in the water column. This transfer of energy from the water column into the benthos fundamentally alters the ecology of the host habitat, resulting in less food/prey available for other filter feeders. At the same time, this may increase resource availability for sturgeon species. Adverse effects of this include reduced body condition, growth, survival, and/or reproduction of native pelagic organisms at the same or similar trophic level as the invader if the native competitor cannot adapt to another food source. These changes would be manifested to a greater or lesser degree up the food chain to higher trophic level organisms in the habitat, including listed salmonids and resident, coastal marine mammals and sea turtles.

In addition to extensive filtration, invasive benthic invertebrates also cause significant physical and chemical changes within their new habitats. As an invasive population grows, it also produces large amounts of feces or pseudofaeces. This provides abundant food sources for bacteria, which supports population explosions of bacteria that exploit this resource. This often leads to anoxia or hypoxia, resulting in wide-scale mortality of benthic inhabitants. Invasive mussel and oyster populations have also slowed flow rates through shallow water habitats and juvenile salmonid rearing areas, leading to increased sedimentation that fills in estuarine habitat.

Introduced diseases and parasites can be expected to impact not only listed species directly, but also species in the food chain on which they rely. Numerous studies identified in the *Response analysis* show how both fishes and invertebrates are susceptible to introduced parasites and diseases, sometimes with devastating levels of mortality as a result. Even moderate levels of morbidity or mortality within a prey or prey base species can lead to less or lesser quality prey available for listed fishes, coastal marine mammals and sea turtles that can in turn lead to reduced body condition, growth, survival, and reproductive resources for listed individuals.

Invasive plants can alter their host environments in a number of ways to the detriment of listed species. Harmful algal blooms resulting for introduced dinoflagellates can not only produce toxins that adversely affect listed individuals directly, but also produce massive die-offs of fish and invertebrate species resulting in less available prey. Estuarine plants, such as *Spartina* spp. and mangroves can rapidly alter the physical environment of an estuary, reducing wave energy, increasing sedimentation, and leading to estuaries filling in that listed fishes rely upon. Plants, such as hyacinth and *C. taxifolia*, can clog waterways also leading to increased sedimentation or outright impairment of fish movement where they have established. Invasive algae on the ocean bottom frequently form a barrier to sedimentation or particle movement into the water column, reducing habitat complexity and, ultimately, the species that can survive within a habitat (potentially prey species of listed organisms or the listed species themselves).

Future Invasive Species Impacts to Listed Species and Critical Habitat

We have established that, regardless of the mechanism by which invasive species have been introduced, they can have a wide range of effects on the environments in which they establish. More importantly, ballast-mediated invasive species have a similar affect on the environment and the species within the environment as do all invasive species generally. It is possible to anticipate the probability of a ballast-mediated invasive species establishing at various ports around the US, but it is impossible to anticipating the specific species that will be established at a specific port and therefore which listed species may be impacted by that specific species. For that reason, NMFS conducted a general assessment of invasion risk from ballast water to determine the probable exposure risk to each of our listed species. The next step is determining the probable response to those exposures.

Some introductions can be beneficial, providing food for listed sea turtles (Russell and Balazs 2009), increasing biodiversity (Nyberg 2007), and establishing new habitat used by native species (Paulo da Cunha Lana 1991, Daehler and Strong 1996, Crooks 1998, Grosholz 2002, Castilla *et al.* 2004, Wallentinus and Nyberg 2007). We have also found instances where the establishment of an invasive has had catastrophic effects on native species and ecosystems, causing high mortality or extinction (Vinogradov *et al.* 1989, Chew 1990, Tsikhon-Lukanina and Reznichenko 1991, Ciguarria and Elston 1997, Zeitsev and Ozturk 2001, Minchin and Rosenthal 2002, Moncheva and Kamburska 2002, Nell 2002, Lleonart *et al.* 2003, Nehring 2006b, Haupt *et al.* 2010, Josefsson and Andersson 2011) or fundamentally altering an ecosystem through a single introduction (Colwell 1996, Castilla *et al.* 2004, Hecky *et al.* 2004, Wallentinus and Nyberg 2007). Far more often, an invasive species usually suppresses an ecologically similar native species low in the food web and the impacts of that invasion have a negligible effect to listed species or their critical habitat.

A thorough review of the literature reveals that just because an invasive species establishes, it is not appropriate to assume there will be an adverse effect to the receiving environment or the listed resources within it. For this reason, we determined what the probability would be that an invasive species establishing in US waters would adversely affect the environment into which it is introduced. We identified three different sources of information to evaluate this question:
(1) evaluations of the impacts of invasive species in terrestrial and freshwater environments written by the US Congress Office of Technology and Assessment (Anttila *et al.* 1998),
(2) impacts of invasive species in the Chesapeake Bay published by Ruiz *et al.* (Ruiz *et al.* 1999), (3) as well as impacts of invasive species in San Francisco Bay (McNickle *et al.* 2001). While none of these studies identified only ballast-mediated invasive species, the *Response Analysis* in this Opinion shows that ballast-mediated invasive species elicit similar responses as other invasive species. Therefore, NMFS expects that any invasive species that are introduced as

a consequence of the USCG's Ballast Water Management Program will have impacts similar to other introduced species with effects ranging from beneficial to harmful.

US Technology Office of Technology and Assessment. This report (Anttila *et al.* 1998) summarized information on non-indigenous species (primarily terrestrial and freshwater) in the United States, as well as what proportion of them were ecologically or economically harmful, beneficial, neutral, or unknown. This non-exhaustive but reasonably comprehensive report used a panel of experts to identify non-indigenous species and the impacts they caused (or failed to cause). In total, 15.3% of non-indigenous species caused "great harm" defined as "significant and difficult-to-control pests of agriculture, rangelands, or forests; seriously foul waterways, irrigation systems, and power plants; cause wide-scale disruption of indigenous ecosystems; or threaten indigenous species with extinction." We believe that all of these categories at least represent species that are competitors, predators, diseases, or habitat modifiers of native species or habitats and would be impactful to the environments into which they were introduced. Overall, 16% were considered harmful, 33% had unknown effects, 26% were considered beneficial, and 25% had both beneficial and harmful effects. The survey included purposefully introduced species of agriculture, hunting, and commercial use, which should upwardly bias the data in the beneficial and beneficial and harmful categories. This is supported by one of the report's findings that only 12% of intentional introductions were considered harmful versus 44% of unintentional introductions. Because this assessment includes terrestrial invasions and intentional introductions, NMFS believes the probability of harm identified in this assessment is the least accurate of the three.

Chesapeake Bay. A comprehensive review of invasive, cryptogenic, and impactful invasive and possibly invasive species was conducted in the Chesapeake Bay by Ruiz *et al.* (Ruiz *et al.* 1999). The authors compiled information on 196 species that were either confirmed as invasive for were cryptogenic (unclear whether they were invasive or native) over the past four centuries. Only cryptogenic species for which significant information existed to support a strong possibility of being invasive were included in these 196 while hundreds of others were excluded. The definition of whether a species was impactful was based exclusively upon available literature that qualitatively or quantitatively ascribed a particular species as being impactful. Additional information on impact for some of these species was available outside of the Chesapeake Bay, but the authors excluded this information from these non-Chesapeake Bay locations because it was possible this information may not apply in the Bay itself. Impacts were assigned to nine categories: competition, habitat change, food-prey, predation, herbivory, hybridization, parasitism, toxicity, or bioturbation which caused "significant and measurable changes in the abundance or distribution of resident species or habitats." This definition, unfortunately, excluded some effects that we would consider significant, such as hybridization or introduced predators for which information was lacking on population-level effects to native prey; this likely causes a slight negative bias in the number of invasives we would consider having an adverse affect. We also expect that the total number of invasives is actually higher than the 196 included, as an unknown subset of those that were excluded are likely actually non-native, causing a positive bias in the estimate. However, we have no mechanism(s) to account for these biases and expect that these caveats do not alter the results so much as to make the estimates of invasive species impacting the Chesapeake Bay unreasonable.

Overall, Ruiz *et al.* (Ruiz *et al.* 1999) found that 39 of 196 (19.9%) invasive or cryptogenic species impacted the ecology of the Chesapeake Bay. Of the remaining species, 51% lacked information on impact and 29% may have an impact but significant information was lacking. Cryptogenic species represented six (15%) of impactful invasives. Based upon this, cryptogenic species compose a relatively small proportion of impactful invasives, but the actual number of impactful invasives is likely significantly higher than the 20% reported here, as a small majority of invasives lacked information to assess their impacts. The ecological categories in which impacts occurred were: competition—69%; habitat change—38%; food/prey—44%; predation—15%; herbivory—21%; hybridization—8%; parasitism—8%; toxicity—0%; and bioturbation—3%. These proportions of impact types roughly correspond with the types of impacts we found globally for invasive species presently in the *Response analysis*.

San Francisco Bay. We also examined available information on the number of invasive species that have established in San Francisco Bay/Delta and the number of these that we found in our information searches that we found to impact the area. The number of invasives, was adopted from Cohen and Carlton (1998). These authors assembled data from the 16th century to the time of publication from published, unpublished, and their personal research on the introduced aquatic organisms of San Francisco Bay/Delta, including some cryptogenic species. The authors identified 234 exotic species as well as 125 cryptogenic species. We expect that some of the cryptogenic species identified by Cohen and Carlton (1998) are actually exotic, but we are not aware of any methodology to further characterize the origin of these species that Cohen and Carlton (1998) have not already done. This proportion may be low, as a 2010 survey in San Francisco Bay identified 92 cryptogenic, of which five were considered exotic and the rest were of unclear origin (Foss 2011). Therefore, in the present analysis, we exclude the 125 cryptogenic species as exotics and accept 234 exotic species as the number of invasive species present. This is roughly corroborated by Foss *et al.* (2007). We expect that this may positively bias our results to some degree, but should be reasonably representative of the number of invasive species present. Furthermore, this assumption would lead to an overestimation of the proportion of invasive species adversely impacting San Francisco Bay/Delta.

The next step is to identify the number of invasive species impacting San Francisco Bay/Delta. To do this, we summed the number of species we identified as impacting this area under the San Francisco Bay case study presented earlier in the *Response analysis*. We also considered the findings of San Francisco Bay invasive species monitoring programs. Based upon our non-exhaustive review in the San Francisco Bay case study, we identified 21 invasive species that have adversely affected native species and/or habitats in San Francisco Bay/Delta. We identified another 22 species negatively impacting this area from Cohen and Moyle (2004), producing a total of 43 adversely affecting invasive species out of 234 (18.4%).

Calculated Rate of Harm. The Congressional study found 15.3% of terrestrial and aquatic invasive species were harmful, while the Chesapeake Bay study found 19.9% of aquatic invasive species were harmful and the San Francisco Bay study identified 18.8% of invasive species as harmful. The main consideration identified by all three methods is the overall lack of information, obligatorily biasing the data in some direction depending on the definition of harm and what has been published on the effects of non-native species. Of these three estimates, NMFS decided to assess potential impacts to listed species and critical habitat using the mean probability of harm (19.9%) identified in the two aquatic studies on the Chesapeake Bay and San Francisco Bay to determine the probability of future ballast-mediated invasive species likely to cause adverse impacts to the environment. This method reflects the most probable rate of invasions likely to cause harm to listed species and their critical habitat. Although each analysis has specific caveats that make each more or less valuable on specific points, we believe that each of these probabilities are relevant and equally significant in determining the likelihood of an invasive species adversely affecting an environment at a global scale.

Estimating Effects of Exposure

In this Opinion, NMFS identified the listed species within the action area that could be affected by an invasive population of organisms. In the *Effects Analysis*, NMFS further identified the probable rate of exposure for these species and the probable effects of that exposure. When an invasive species has established, we mean that it has formed one or more reproductive population(s) in a host location and generally occur throughout the year. Although listed species would generally co-occur with the invasive species in time and space, the requirement for co-occurrence in space and time is not a prerequisite, as invasive species can have indirect effects to habitat (such as alteration of physical conditions) and that impact can have a significant, if not primary impact to listed species. While it is not possible to anticipate the exact species that will invade a particular location, it is possible to anticipate the probability of an establishment. Through the Response Analysis, we found that likely responses of listed species to future invasions would be expected to be similar to past invasions, which can be diverse (including competition for prey/nutrition and space, predation, disease, genetic hybridization, and habitat alteration) and not necessarily negative (such as providing additional prey resources).

Having established that the effects range from beneficial to harmful, NMFS estimated the proportion of invasive species that have been harmful using several hundred invasive species in the analysis to ensure there was very little chance of there being an impact we had not considered in our analysis. Furthermore, with the range of harmful impact rates being between 18.8% and 19.9% in the two aquatic analyses we conducted, NMFS is confident this is the rate of harmful impacts that will be seen in the future, although we cannot predict which species will establish in the future and the specific effects that those species would have. By using the mean rate of harmful impacts, NMFS can assess the adverse impacts of this action in the most probable way. However, because listed species and critical habitats often occur over multiple bioregions, to determine the effects to listed species and critical habitat for the conclusions of this Opinion, we must summarize the information from this section in the next section. While the probability of an invasive species establishing in a specific bioregion may be negligible, the cumulative effect of those probabilities to listed species and critical habitats may be significant.

Furthermore, in order to determine the probability of an invasive species establishing through time, NMFS estimated when these numerical standards will be required based on the history of ballast water regulations. NANPCA was initially passed in 1990. After six years, that law was updated and NISA was passed. After eight years of working under the initial framework established by NISA, the USCG issued a final rule that mandated the use of BWE in 2004. And after eight years of mandatory BWE, the USCG is again reducing the probability of ballast-mediated invasions by introducing a numerical standard. Given the history of regulation change, it is not unreasonable to anticipate another regulation change in seven to eight years. Although past performance shows us that changes may well occur eight years from present, no change is required in eight years and we would run the risk of underestimating exposure at the proposed numeric standard if we limited our review to eight years. However, given the history of this program, NMFS anticipates within a duration of two historic cycles (16 years) there will be program differences not reflected in the assumptions of this Opinion (Table 42). We therefore do not consider effects beyond this timeframe. We are aware that effects of invaders now and within the next decade and a half will extend beyond this period, but have no information available to use to determine what, where, when, and how those effects will impact listed resources and their critical habitats. As such, any analysis of these effects would be arbitrary.

Table 42. Probabilities of ballast-mediated invasions in each bioregion annually as well as over the next 16 years.

Bioregion	Approximate Discharge Volume (tonnes)	Probability of Annual Invasion	Probability of Invasion over 16 Years	Probability of Harm	Probability of Harm over 16 Years
NA-S1	32000	0.0017	0.0269	0.0003	0.0053
NEP-I	8,540,000	0.4558	0.9999	0.0882	0.7718
NEP-II	603,000	0.032	0.4057	0.0062	0.0947
NEP-IV	1,810,000	0.0964	0.8025	0.0187	0.2607
NEP-V	6,540,000	0.3491	0.9991	0.0675	0.6731
NEP-VI	8,890,000	0.4745	0.9999	0.0918	0.7858
CAR-I	94,010,000	1.0	1.0	0.9707	1.0
CAR-IV	10,190,000	0.544	1.0	0.1053	0.8314
NA-ET2	579,000	0.0309	0.3948	0.006	0.0918
NA-ET3	25,560,000	1.0	1.0	0.2639	0.9926
CAR-VII	3,650,000	0.1946	0.9687	0.0377	0.4593
SP-IX	5,679	0.0003	0.0048	0.00005	0.0007
SP-XII	62,433	0.0033	0.0515	0.0006	0.0096
SP-XXI	507,000	0.0271	0.3557	0.0052	0.0800

Probability of Harmful Invasions by Bioregion

Ballast Water Bioregion NA-S1. NA-S1 includes all ports along the northern coast of Alaska. Within this bioregion, newly establishing invasive species have the potential to impact southern DPS spotted seal.

Between 2005 and 2010 in NA-S1, the corrected average ballast water discharge volume was approximately 32,000 tonnes. The reported volume was corrected to account for incomplete reporting. If this five year discharge rate is similar for the next 16 years, NMFS would expect the probability for a ballast-mediated invasion in this bioregion to be approximately 0.0017 per year under the numerical standard. Assuming a 16 year timeframe before another rulemaking introduces a new management plan, the probability of an invasion in this bioregion is approximately 0.0269. Additionally, the probability of a ballast-mediated invasion causing harm to a listed species would be approximately 0.0003 per year but 0.0053 over 16 years.

Ballast Water Bioregion NEP-I. NEP-I includes all ports of western Alaska and around the southern coast to Juneau. Within this bioregion, newly establishing invasive species have the potential to impact southern DPS spotted seal and western DPS Steller sea lions.

Between 2005 and 2010 in NEP-I, the corrected average ballast water discharge volume was approximately 8.54×10^6 tonnes. The reported volume was corrected to account for incomplete reporting. If this five year discharge rate is similar for the next 16 years, NMFS would expect the probability for an invasion in this bioregion to be approximately 0.4558 per year under the numerical standard. Assuming a 16 year timeframe before another rulemaking introduces a new management plan, the probability of an invasion in this bioregion is approximately 0.9999. Additionally, the probability of a ballast-mediated invasion causing harm to a listed species would be approximately 0.0882 per year but 0.7718 over 16 years.

Ballast Water Bioregion NEP-II. NEP-II includes ports of the Aleutian Islands including Dutch Harbor and Cook Inlet. Within this bioregion, newly establishing invasive species have the potential to impact Cook Inlet DPS beluga whale, southern DPS spotted seal, and western DPS Steller sea lions and their critical habitat.

Between 2005 and 2010 in NEP-II, the corrected average ballast water discharge volume was approximately 603,000 tonnes. The reported volume was corrected to account for incomplete reporting. If this five year discharge rate is similar for the next 16 years, NMFS would expect the probability for an invasion in this bioregion to be approximately 0.032 per year under the numerical standard. Assuming a 16 year timeframe before another rulemaking introduces a new management plan, the probability of an invasion in this bioregion is approximately 0.4057. Additionally, the probability of a ballast-mediated invasion causing harm to a listed species would be approximately 0.0062 per year but 0.0947 over 16 years.

Ballast Water Bioregion NEP-IV. This bioregion covers the coasts of Oregon and Washington as well as ports in the Columbia River and Puget Sound. Within this bioregion, new non-native species have the potential to impact SONCC coho salmon and their critical habitat, lower Columbia River coho salmon, Oregon coast coho salmon and their critical habitat, Pacific eulachon and their critical habitat, lower Columbia River Chinook salmon and their critical habitat, upper Columbia River spring-run Chinook salmon and their critical habitat, Snake River fall run Chinook salmon and their critical habitat, Snake River spring/summer run Chinook salmon and their critical habitat, upper Willamette River Chinook salmon and their critical habitat, Hood Canal summer run chum salmon and their critical habitat, Columbia River chum salmon and their critical habitat, southern DPS green sturgeon and their critical habitat, Snake River sockeye salmon and their critical habitat, Ozette Lake sockeye salmon and their critical habitat, Puget Sound steelhead, Snake River basin steelhead and their critical habitat, upper Columbia River steelhead and their critical habitat, middle Columbia River steelhead and their critical habitat, lower Columbia River steelhead and their critical habitat, upper Willamette River steelhead and their critical habitat, Puget Sound/Georgia Basin DPS yelloweye rockfish, Puget Sound/Georgia Basin DPS canary rockfish, Puget Sound/Georgia Basin DPS bocaccio, leatherback sea turtles and their critical habitat, eastern DPS Steller sea lion and their critical habitat, Southern resident killer whale and their critical habitat, and humpback whales.

Between 2005 and 2010 in NEP-IV, the corrected average ballast water discharge volume was approximately 1.81×10^6 tonnes. The reported volume was corrected to account for incomplete reporting. If this five year discharge rate is similar for the next 16 years, NMFS would expect the probability for an invasion in this bioregion to be approximately 0.0964 per year under the numerical standard. Assuming a 16 year timeframe before another rulemaking introduces a new management plan, the probability of an invasion in this bioregion is approximately 0.8025. Additionally, the probability of a ballast-mediated invasion causing harm to a listed species would be approximately 0.0187 per year but 0.2607 over 16 years.

Ballast Water Bioregion NEP-V. This bioregion covers northern California including the major ports of SF Bay. Within this bioregion, newly establishing invasive species have the potential to impact white abalone, black abalone and their critical habitat, California coastal Chinook salmon and their critical habitat, central valley spring-run Chinook salmon and their critical habitat, Sacramento River winter-run Chinook salmon and their critical habitat, central California coast coho salmon and their critical habitat, southern Oregon/northern California coast (SONCC) coho salmon and their critical habitat, green sturgeon and their critical habitat, Pacific eulachon and their critical habitat, central California coast steelhead and their critical habitat, northern California steelhead and their critical habitat, south-central California coast steelhead and their critical habitat, California central valley steelhead and their critical habitat, leatherback sea turtles and their critical habitat, north Pacific Ocean DPS loggerhead sea turtles, eastern DPS Steller sea lions and their critical habitat, and humpback whales.

Between 2005 and 2010 in NEP-V, the corrected average ballast water discharge volume was approximately 6.54×10^6 tonnes. The reported volume was corrected to account for incomplete reporting. If this five year discharge rate is similar for the next 16 years, NMFS would expect the probability for an invasion in this bioregion to be approximately 0.3491 per year under the numerical standard. Assuming a 16 year timeframe before another rulemaking introduces a new management plan, the probability of an invasion in this bioregion is approximately 0.9991. Additionally, the probability of a ballast-mediated invasion causing harm to a listed species would be approximately 0.0675 per year but 0.6731 over 16 years.

Ballast Water Bioregion NEP-VI.

Bioregion NEP-VI includes all ports from southern California south along the Mexican Coast, though this Opinion only considers the ports in southern California (Long Beach, Los Angeles, and San Diego). Within this bioregion, newly establishing invasive species have the potential to impact southern California steelhead and their critical habitat, eastern DPS Steller sea lions and their critical habitat, green sea turtles, leatherback sea turtles and their critical habitat, North Pacific DPS loggerhead sea turtles, olive ridley sea turtles, humpback whales.

Between 2005 and 2010 in NEP-VI, the corrected average ballast water discharge volume was approximately 8.89×10^6 tonnes. The reported volume was corrected to account for incomplete reporting. If this five year discharge rate is similar for the next 16 years, NMFS would expect the probability for an invasion in this bioregion to be approximately 0.4745 per year under the numerical standard. Assuming a 16 year timeframe before another rulemaking introduces a new management plan, the probability of an invasion in this bioregion is approximately 0.9999.

Additionally, the probability of a ballast-mediated invasion causing harm to a listed species would be approximately 0.0918 per year but 0.7858 over 16 years.

Ballast Water Bioregion CAR-I. Bioregion CAR-I includes all US ports in the entire Gulf of Mexico and around the southern tip of Florida. Within this bioregion, newly establishing invasive species have the potential to impact gulf sturgeon and their critical habitat, green sea turtle, hawksbill sea turtle, Kemp's ridley sea turtle, leatherback sea turtles, Northwest Atlantic Ocean DPS loggerhead sea turtles, elkhorn coral and their critical habitat, staghorn coral and their critical habitat, and Johnson's sea grass and their critical habitat.

Between 2005 and 2010 in CAR-I, the corrected average ballast water discharge volume was approximately 94.01×10^6 tonnes. The reported volume was corrected to account for incomplete reporting. If this five year discharge rate is similar for the next 16 years, NMFS would expect approximately 5.01 ballast-mediated invasive species to establish each year, which equates to a probability of 1.0. Assuming a 16 year timeframe before another rulemaking introduces a new management plan, the probability of an invasion in this bioregion is 1.0. Additionally, the probability of a ballast-mediated invasion causing harm to a listed species would be approximately 0.9707 per year but 1.0 over 16 years.

Ballast Water Bioregion CAR-IV. CAR-IV includes all ports around the US Virgin Islands and Puerto Rico. Within this bioregion, newly establishing invasive species have the potential to impact green turtles and critical habitat, Hawksbill turtle and their critical habitat, leatherback sea turtles and their critical habitat, Northwest Atlantic Ocean DPS loggerhead sea turtles, olive ridley sea turtles, smalltooth sawfish, elkhorn coral and their critical habitat, and staghorn coral and their critical habitat.

Between 2005 and 2010 in CAR-IV, the corrected average ballast water discharge volume was approximately 10.19×10^6 tonnes. The reported volume was corrected to account for incomplete reporting. If this five year discharge rate is similar for the next 16 years, NMFS would expect the probability for an invasion in this bioregion to be approximately 0.544 per year under the numerical standard. Assuming a 16 year timeframe before another rulemaking introduces a new management plan, the probability of an invasion in this bioregion is approximately 0.9999. Additionally, the probability of a ballast-mediated invasion causing harm to a listed species would be approximately 0.1053 per year but 0.8314 over 16 years.

Ballast Water Bioregion NA-ET2. NA-ET2 includes all ports from Maine to Massechusetts. Within this bioregion, newly establishing invasive species have the potential to impact Gulf of Maine, Atlantic salmon, Gulf of Maine DPS of Atlantic sturgeon, New York Bight DPS of Atlantic sturgeon, Chesapeake Bay DPS of Atlantic sturgeon, Carolina DPS of Atlantic sturgeon, South Atlantic DPS of Atlantic sturgeon, shortnose sturgeon, Kemp's ridley sea turtle, leatherback sea turtles, and Northwest Atlantic Ocean DPS loggerhead sea turtles.

Between 2005 and 2010 in NA-ET2, the corrected average ballast water discharge volume was approximately 579,000 tonnes. The reported volume was corrected to account for incomplete reporting. If this five year discharge rate is similar for the next 16 years, NMFS would expect the probability for an invasion in this bioregion to be approximately 0.0309 per year under the numerical standard. Assuming a 16 year timeframe before another rulemaking introduces a new management plan, the probability of an invasion in this bioregion is approximately 0.3948. Additionally, the probability of a ballast-mediated invasion causing harm to a listed species would be approximately 0.006 per year but 0.0918 over 16 years.

Ballast Water Bioregion NA-ET3. NA-ET3 includes all ports south of Massechusetts to the southern edge of the Chesapeake Bay. Within this bioregion, newly establishing invasive species have the potential to impact Gulf of Maine DPS of Atlantic sturgeon, New York Bight DPS of Atlantic sturgeon, Chesapeake Bay DPS of Atlantic sturgeon, Carolina DPS of Atlantic sturgeon, South Atlantic DPS of Atlantic sturgeon, shortnose sturgeon, green sea turtles, hawksbill sea turtles, Kemp's ridley sea turtle, leatherback sea turtles, and Northwest Atlantic Ocean DPS loggerhead sea turtles.

Between 2005 and 2010 in NA-ET3, the corrected average ballast water discharge volume was approximately $25.56\text{x}10^6$ tonnes. The reported volume was corrected to account for incomplete reporting. If this five year discharge rate is similar for the next 16 years, NMFS would expect approximately 1.364 new ballast-mediated invasive species per year, which equates to a probability of 1.0. Assuming a 16 year timeframe before another rulemaking introduces a new management plan, the probability of an invasion in this bioregion is approximately 1.0. Additionally, the probability of a ballast-mediated invasion causing harm to a listed species would be approximately 0.2639 per year but 0.9926 over 16 years.

Ballast Water Bioregion CAR-VII. Bioregion CAR-VII includes all ports between northern Florida and North Carolina. Within this bioregion, newly establishing invasive species have the potential to impact Gulf of Maine DPS of Atlantic sturgeon, New York Bight DPS of Atlantic sturgeon, Chesapeake Bay DPS of Atlantic sturgeon, Carolina DPS of Atlantic sturgeon, South Atlantic DPS of Atlantic sturgeon, shortnose sturgeon, green sea turtles, hawksbill sea turtles, Kemp's ridley sea turtle, leatherback sea turtles, Northwest Atlantic Ocean DPS loggerhead sea turtles,

Between 2005 and 2010 in CAR-VII, the corrected average ballast water discharge volume was approximately $3.65\text{x}10^6$ tonnes. The reported volume was corrected to account for incomplete reporting. If this five year discharge rate is similar for the next 16 years, NMFS would expect the probability for an invasion in this bioregion to be approximately 0.1946 per year under the numerical standard. Assuming a 16 year timeframe before another rulemaking introduces a new management plan, the probability of an invasion in this bioregion is approximately 0.9687. Additionally, the probability of a ballast-mediated invasion causing harm to a listed species would be approximately 0.0377 per year but 0.4593 over 16 years.

Ballast Water Bioregion SP-IX. Bioregion SP-IX includes all ports of American Somoa. Within this bioregion, newly establishing invasive species have the potential to impact green sea turtles, hawksbill sea turtles, leatherback sea turtles, south Pacific DPS loggerhead sea turtles, olive ridley sea turtles, and humpback whales.

Between 2005 and 2010 in SP-IX, the corrected average ballast water discharge volume was approximately 5,679 tonnes. The reported volume was corrected to account for incomplete reporting. If this five year discharge rate is similar for the next 16 years, NMFS would expect the probability for an invasion in this bioregion to be approximately 0.0003 per year under the numerical standard. Assuming a 16 year timeframe before another rulemaking introduces a new management plan, the probability of an invasion in this bioregion is approximately 0.0048. Additionally, the probability of a ballast-mediated invasion causing harm to a listed species would be approximately 0.00005 per year but 0.0007 over 16 years.

Ballast Water Bioregion SP-XII. Bioregion SP-XII includes all the ports of Guam. Within this bioregion, newly establishing invasive species have the potential to impact green sea turtles, hawksbill sea turtles, leatherback sea turtles, south Pacific DPS loggerhead sea turtles, olive ridley sea turtles, and humpback whales.

Between 2005 and 2010 in SP-XII, the corrected average ballast water discharge volume was approximately 62,433 tonnes. The reported volume was corrected to account for incomplete reporting. If this five year discharge rate is similar for the next 16 years, NMFS would expect the probability for an invasion in this bioregion to be approximately 0.0033 per year under the numerical standard. Assuming a 16 year timeframe before another rulemaking introduces a new management plan, the probability of an invasion in this bioregion is approximately 0.0515. Additionally, the probability of a ballast-mediated invasion causing harm to a listed species would be approximately 0.0006 per year but 0.0096 over 16 years.

Ballast Water Bioregion SP-XXI. Bioregion SP-XXI is an isolated area in the center of the Pacific Ocean surrounding the islands of Hawaii. Within this bioregion, newly establishing invasive species have the potential to impact Hawaiian monk seal and their critical habitat, green sea turtles, hawksbill sea turtles, leatherback sea turtles, North Pacific DPS loggerhead sea turtles, olive ridley sea turtles, and humpback whales.

Between 2005 and 2010 in SP-XXI, the corrected average ballast water discharge volume was approximately 507,000 tonnes. The reported volume was corrected to account for incomplete reporting. If this five year discharge rate is similar for the next 16 years, NMFS would expect the probability for an invasion in this bioregion to be approximately 0.0271 per year under the numerical standard. Assuming a 16 year timeframe before another rulemaking introduces a new management plan, the probability of an invasion in this bioregion is approximately 0.3557. Additionally, the probability of a ballast-mediated invasion causing harm to a listed species would be approximately 0.0052 per year but 0.08 over 16 years.

Ballast Water Discharges to the coastal US. While there are not NMFS protected resources in all waters of the US, there are NMFS species in all marine waters. Between 2005 and 2010, the corrected average ballast water discharge volume was approximately 182.6×10^6 tonnes. The reported volume was corrected to account for incomplete reporting. If this five year discharge rate is similar for the next 16 years, NMFS would expect approximately 9.7 new ballast-mediated invasive species per year under the numerical standard. Therefore, new invasive species are still expected to establish in waters of the US annually, but at a reduced rate compared to BWE. Additionally, given the rate of invasions that is expected to harm listed species, approximately 1.877 harmful invasions will occur each year in the US.

EFFECTS OF THE PROPOSED ACTION: EFFECTS OF THE BALLAST WATER MANAGEMENT PROGRAM

Overview of the Program

Adaptive Management from Monitoring and Enforcement

The USCG ballast water program has been functioning for over 20 years. While this is the first time the USCG has chosen to have NMFS and USFWS consult on the potential impacts of its program to listed species and their critical habitat, there are examples of how successful the ballast water management program monitoring has been in the past. Furthermore, the USCG has other programs with similar monitoring plans that have also proven successful. These monitoring efforts have led to changes within the programs being monitored, showing that the monitoring in place is not only effective but also led to changes within the programs to make the programs more successful.

Voluntary BWE to mandatory BWE. When NISA was passed in 1996, the USCG implemented voluntary BWE as a practicable ballast water treatment technique. However, there were several conditions that would allow vessels to discharge ballast water at a port without having conducted BWE. When the USCG evaluated the voluntary program in a 2002 report to Congress, the report concluded that low participation in the voluntary program resulted in insufficient data for an accurate assessment of its effectiveness. This finding triggered the requirement in NISA that the voluntary ballast water management program become mandatory. Two years later in July 2004, the USCG published the final rule making BWE mandatory, requiring all vessels equipped with ballast water tanks and bound for ports or places of the United States to conduct a mid-ocean BWE, retain their ballast water onboard, or use an alternative environmentally sound BWM method approved by the Coast Guard.

Also in 2004, the USCG published penalties for failure to comply with the reporting requirements located in 33 CFR 151 and broadened the applicability of the reporting and recordkeeping requirements to a majority of vessels bound for ports or places of the United States. The penalties related to record keeping are another deterrent to vessels considering avoiding ballast water management actions. Since being passed eight years ago, several criminal cases have been initiated related to record keeping.

Successful Ballast Water Monitoring and Enforcement. The USCG monitors every ship that enters US waters annually. This amounts to approximately 10,000 foreign vessels and 12,000 domestic vessels. After BWE was made mandatory, the USCG was able to test the salinity of the water to determine whether open ocean exchange had occurred.

The primary means of monitoring and enforcement is to analyze the records vessels are required to maintain. When the USCG boards a vessel, they analyze many different aspects of ship safety and statutory requirements, including ballast water treatment compliance. To determine whether ballast water has been treated, the USCG will look at the record book to make sure all treatments were recorded, assess the crew’s ability to operate the treatment device, and inspect the treatment equipment to determine if it is being maintained. This cursory inspection leads to more thorough inspections if it is apparent the equipment is not being used. Vessels that have been cited for violations previously will receive more thorough investigations of their equipment to ensure they are no longer in violation of the USCG's ballast program. In the event violations are discovered, the USCG will initiate criminal prosecution of the vessels. The USCG is currently in the midst of a prosecution for presenting false information in the ballast water record book to USCG inspectors. The false statement charge was for the crew's attempt to conceal the fact that fuel oil was leaking into the forepeak ballast tank.

Oil-water separator monitoring and enforcement. The oil water separator program is similar to the proposed ballast water monitoring program in that every vessel is boarded at least once a year and as part of the Act for Preventing Pollution from Ships monitoring. While the monitoring goals and methods of the programs may be different, both rely on inspecting record books and physically sampling the vessel.

In calendar year 2011, the USCG has uncovered 18 instances of criminal actions relating to oil water separator inspections. This is down from 22 criminal prosecutions in 2010. In addition, the USCG has initiated two civil penalty cases for oil water separator-related incidents resulting in one unproven case and one still under review by the USCG Hearing Officer. The proven case resulted in a civil penalty of $13,000 and the one under review is requesting a penalty of $19,000. Finally, USCG inspectors issued 268 deficiencies for oil water separator equipment in calendar year 2011 with the large majority of those being rectified during the course of the inspection. The oil water separator program accounted for a small fraction (0.70%) of the total vessel related deficiencies identified on vessels related to other programs in 2011 (38,568).

Evaluation of Those Components of the Ballast Water Management Program

Assessment of the Program

The USCG requested programmatic consultation with NMFS, which will result in NMFS determining the effect of the program on listed species and their critical habitat as opposed to NMFS determining the effect of the action of that program on listed species or critical habitat. This distinction is important because many programs authorize actions with adverse effects that cannot be further mitigated. When that occurs, the programs include feedback loops and adaptive management that allow the effects to listed species and their critical habitat to be reduced promptly in the future after further mitigation is developed.

Ballast water standards are an example of an authorized activity under the USCG's ballast water management program that are likely to have adverse effects to listed species and their critical habitat without any further mitigation being available at this time. As previously noted, the probability of an invasion occurring depends on: (1) the volume of ballast water discharged and (2) the number of non-native organisms in that discharge. If no ballast water is discharged there is no exposure, or if no organisms are released in a ballast water discharge there is no exposure. Any other scenario has varying levels of affect depending on those two factors.

Under the NISA of 1996, the USCG is tasked with preventing invasive species from entering US waters; however they are tasked with doing so practicably. Therefore, the USCG is required to consider economic and environmental impacts of any method to prevent an invasion. It is not economically practicable to eliminate ballast water discharges. And it is not technologically possible to eliminate all living organisms from ballast water discharge without considerable environmental impacts. Therefore, while it appears the action being proposed by the USCG may adversely affect listed species and their critical habitat, the actual questions NMFS must address are whether their proposal is the best practicable solution to invasive species, whether the program allows for the lowest probability of a new invasion occurring given the current state of technology, and whether there is anything the USCG could do to further protect listed species and their critical habitat that they are not currently proposing to do.

Many action agencies request consultation on a number of small actions with relatively minor effects that all fall under a single national level program. In many cases, programs authorize activities that could be described as "death by one thousand cuts;" allowing many very small projects with small adverse impacts to occur without determining whether those small projects could cumulately have large impacts. For instance, if the USCG requested consultation with NMFS on each individual vessel's ballast water management, while cumbersome under the ESA, the cumulative effects analysis should identify the impacts of all of those smaller actions, which may appear to have an individually negligible impact to listed species or their critical habitat.

Therefore, the analytical process of assessing the function of a program is different than analyzing the effects of an individual or collection of projects and in many ways, requires more of a long-term approach. Analyzing the USCG's program allows NMFS to take into consideration the best available technology, acknowledge its imperfections, analyze the legal framework of the USCG's program, and reach a conclusion about the program instead of the cumulative effect of the individual proposed discharges. NMFS uses the same consultation criteria to assess all programs, including the USCG's ballast water management program. In the Approach to the Assesssment section, NMFS identifies seven criteria that the USCG's program would need to propose within their program, demonstrate an ability to achieve and monitor those aspects of their program, and report to NMFS their success in order to insure their program is not likely to jeopardize listed species or adversely modify their critical habitat.

USCG's Ballast Water Program's Ability to Estimate the Individual and Cumulative Effects of Ballast Water Discharges

The USCG has monitored ballast water for over two decades. Because the USCG authority for managing ballast water comes from the NISA, they are routinely assessing the impacts of invasive species. To assess invasive species impacts, the USCG gathers information on ballast water through reporting, record keeping, and monitoring. In addition to program monitoring activities, the USCG also reviews invasive species literature and continues, as mandated by NISA, to review ballast water treatment performance to reduce individual and cumulative impacts from ballast water discharge. While the proposed action is the best numerical standard that is technologically achievable at this time, the USCG has implemented practicability and periodic reviews with the goal of identifying technology that can practicably achieve a more stringent ballast water standard in order to further reduce the probability of an invasive species establishing.

The probability of an invasive species establishing depends on two variables. The first, the number of organisms discharged per volume of water, is regulated by the USCG and the USCG has proposed the most stringent discharge standard that is technologically achievable. The second, the total volume of ballast water discharged, cannot be regulated by the USCG because NISA does not give them that authority. Although we predict a probability of invasion(s), we cannot anticipate which species would compose these invasions.

The USCG, as mandated by NISA, is required to limit the threat of invasive species as much as is practicable. To achieve this mandate, the USCG has opted to identify the most stringent numerical standard that is achievable, which is the subject of its recent FR. However, eight states have established more stringent ballast water numeric standards than the USCG is proposing. Of those eight, California has proposed the most stringent standard of any state and New York has proposed the most stringent standard on the East Coast. Additionally, EPA is proposing essentially the same standards to be achieved under their Vessel General Permit. One major difference between the Vessel General Permit and the USCG standard is that the US Congress exempted certain vessel types under NISA but did not exempt them from discharges under the Clean Water Act, so more vessels will be required to meet the same national standards under the Vessel General Permit than under the Ballast Water Management Program. Additionally, the USCG standards evaluate whether a specific technology can be installed on all types of vessels or whether it would only fit some vessel types and designs, whereas, the Vessel General Permit does not take into account the achievability of the standard on various vessel types.

Those states with more stringent standards have allowed for onshore discharge, no discharge, or discharging potable water just like the USCG; however, if ballast water is to be discharged, vessels in those states must meet a more stringent standard. The manner in which the numeric standards for those states was derived is different than the way the USCG determined the national standard. In all cases, evaluations of current technology were conducted but for instance in California, a technology is considered potentially achievable, and therefore acceptable as a treatment system, if it meets the standard just once out of all samples taken. Of 46 technologies in California, only eight were potentially able to achieve a more stringent standard and none of the technologies were capable of meeting a more stringent standard in every test. Only three technologies tested in California have proven to be able to meet the more stringent standard more than 50% of the time. While the USCG rule is not numerically as stringent, the standard set will be met during every test to be cleared for use in US waters. Additionally, a group of Great Lakes states has voiced concern about the ability for technology to consistently achieve California’s more stringent standard. Even California is concerned about the long-term goals of its ballast water standard of zero organisms by 2020 and have established a review in 2016 that will assess whether that standard is possible.

Like California, New York has established standards 1000 times more stringent than those proposed by the USCG. However, because no technology exists to meet the standards introduced by the state of New York in 2008, in February 2012, the state extended the deadline to meet the New York standard, meaning until the date that technology is capable of meeting a standard 1000 times more stringent than the USCG’s, New York effectively has no ballast water standard.

Based on sampling conducted by foreign nations, it appears there is technology available that can consistently meet the standard proposed by the USCG. Furthermore, because it appears technology is not yet capable of achieving a more stringent standard, the USCG monitors the total ballast water discharge through their reporting and record keeping requirements. Additionally, the USCG has established periodic reviews every three years with the intention of making the ballast water standard more stringent if practical technology is discovered.

USCG's Ability to Estimate Stressors Resulting from the Ballast Water Discharges

Many publications have attempted to predict which non-native species are likely to invade and what their impacts will be (Guisan and Thuiller 2005, Vaclavik and Meentemeyer 2009)(Williamson 1996, Williamson 1999, Enserink 1999, Peterson and Vieglais 2001, Kolar and Lodge 2001), but those same studies have determined that it is not currently possible to identify individual non-native species with the highest probability of invading an area next, nor is it possible to anticipate specific locations those species may establish viable populations. Because it is not possible to predict which species are likely to become established in a given area or what the anticipated impacts of an invasion could be, the USCG and NMFS have determined that the range of ecological effects documented over the past 50 years of invasive species science are likely to be similar to the ecological effects of future invasions.

Given the unpredictability of invasive species establishment, the USCG has proposed a numerical standard that will limit, to the maximum extent practicable, the probability of an invasion occurring. Furthermore, the USCG acknowledges that the total volume discharged is critical, but the agency has no authority to regulate the volume of ballast water discharged. Because of this, the USCG requires reporting and record keeping that monitor all aspects of ballast water from its uptake to discharge. The USCG has also implemented three year practicability and periodic reviews to evaluate the practicability of new technologies being developed with the goal of moving the numerical ballast water standard from the current standard to one that is more stringent, ultimately eliminating the threat of invasive species.

USCG's Ability to Estimate Compliance with the Ballast Water Management Program

The USCG has established monitoring, reporting, and record keeping requirements of all vessels to ensure that all vessels and vessel operators have complied with the conditions, restrictions, and mitigation measures required by the proposed action. The records are checked when monitoring takes place; the USCG boards every vessel that enters US waters at least once each year. As previously described, a vessel is required to report ballast water discharges prior to making them and maintain a record for at least 24 months of information about the ballast water in the tanks, treatment, and discharge. Furthermore, mitigation measures are established by the FR and while reporting and record keeping are not specifically required to verify the vessels have conducted the mitigation, the reports and records will identify the location of ballast discharge to ensure vulnerable areas are avoided and protected. The FR also contains specific compliance monitoring to ensure that all vessels have been equipped with ballast water treatment technology that was approved by the USCG and that the technology is being maintained and continues to meet the numeric standard. In the event that non-compliance is identified, the NISA provides tough enforcement regulations that allow the USCG to bring charges against the vessel operators. Under the similar oil water separator monitoring program, approximately 20 criminal charges are discovered every year and hundreds of other violations are documented and violating vessels fined. While it is more difficult to prove a vessel is not complying with BWE, the USCG has brought two cases related to record keeping and BWE. Under the proposed numerical ballast water standard, the USCG expects monitoring water and sediment samples to lead to increased compliance and easier enforcement.

USCG's Ability to Determine Exposures of Listed Resources to the Effects of the Proposed Action

The USCG has compliance monitoring, reporting, and record keeping requirements that allow them to reliably estimate whether and to what degree specific endangered or threatened species and their critical habitat are likely to be exposed to the direct or indirect effects of the discharges to be authorized by the proposed action. The reporting and record keeping required by each vessel is maintained at the National Ballast Information Clearinghouse. Because every vessel is boarded at least once a year and those vessels are required to report their ballast information electronically, the NBIC data is kept up to date annually. Based on years of records, the USCG is well-informed of the reporting rates, the reporting reliability, and the volume of ballast water discharged annually. Furthermore, annual compliance monitoring ensures that vessels are equipped with ballast water treatment technology and that those technologies continue to meet the ballast water standard after being installed.

Based on the mandatory monitoring requirements in the FR, the USCG can monitor the volume of treated ballast water being discharged to various ports, estuaries, and marine bioregions to determine the listed species in the area that may be affected by the proposed ballast water discharges. The goal of the USCG, as authorized by NISA, is to reduce the risk of invasive species from ballast water discharge as much as is possible. Reducing the risk of invasive species is accomplished in one of two ways: (1) reducing the number of organisms per volume of discharge or (2) reducing the volume of ballast water discharged. The USCG does not have the authority to manage the volume of ballast water discharged and therefore can only reduce the number of organisms per discharge to the maximum extent practicable. Furthermore, they currently monitor the volume of ballast water discharged by each ship, while also conducting practicability reviews and periodic reviews to identify new technology capable of achieving a more stringent standard. In these ways, the USCG is able to ensure the exposure of listed resources to ballast water discharge is not greater than has been analyzed in this Opinion. All these monitoring efforts will allow the USCG to understand the ballast water discharges and the potential exposure of listed species to ballast water discharges and undertake adaptive management measures if problems are identified in their program.

USCG's Ability to Monitor Adverse Effects from Authorized Activities

As described above, it is not possible to predict which species may invade or what those impacts may be. As a result, NMFS and USCG have determined that the likely effects of new listed species will be similar to the effects seen by the 500 non-native species currently inhabiting the waters of the US. Furthermore, there is generally a delay between when a species is introduced and when its population is first identified as being invasive, the point at which adverse effects could potentially occur. This is because, as the final PEIS shows, there is a high likelihood that most species that are introduced will go extinct rather than establish in their new environment.

Ballast water is not the only source of non-native species in waters of the US. In fact, Molner *et al.* (2008) identify all sources of invasive species and identify shipping as responsible for no more than 69% of total invasions, and of that number, ballast water accounts for between 31 and 61% of shipping related invasive species. Therefore, simple environmental monitoring would not be able to detect whether invasions were resulting from this proposed action or from one of the many more numerous sources of invasive species.

To address the uncertainties about non-native species and their effects, the USCG has implemented compliance monitoring and effectiveness monitoring that will ensure ballast water treatment technology has been installed in all vessels and is effectively meeting the proposed numerical standard. That coupled with monitoring the total volume of ballast water discharge allows the USCG to monitor the risk of new invasive species resulting from ballast water discharge and to anticipate any adverse effects caused by those invasions. All of these monitoring efforts will allow the USCG to undertake adaptive management measures if problems are identified in their program.

USCG's Consideration of Species' Status and Population Effects

The USCG began working with NMFS and USFWS in 2006 to evaluate the effects of a ballast water standard on listed species and their critical habitat. The USCG's primary intent during this process was reducing the risk of invasive species establishing and therefore the potential adverse effects of invasive species. While this new numerical standard is a clear improvement over BWE, both in terms of reducing the risk of invasion and increasing the enforceability of ballast water regulations, there is still a potential risk to listed species.

The USCG proposed the most stringent practicable ballast water management solution to protect the environment. Furthermore, the USCG program includes a practicability review and later periodic reviews that allow the USCG to increase the stringency of the standard as soon as technology is developed that can practicably achieve a greater stringency. Additionally, the USCG included three alternative discharge techniques that are acceptable: (1) no ballast discharge, (2) discharge to an on-site facility or barge, and (3) using potable water in ballast tanks. The USCG expects that some vessels may decide not to discharge ballast water due to the increased risk of enforcement and cost of installing treatment technologies; however, most vessels are unable to operate safely without taking on and discharging ballast water. There are currently no on-site discharge facilities in the US, but some ports may choose to develop discharge sites. And using potable ballast water, while an option, may increase the strain on domestic drinking water resources and have an adverse economic impact.

In addition to the periodic and practicability reviews, the USCG will also conduct compliance monitoring to ensure all vessels are equipped with treatment technologies and that those technologies are being used. And as part of those reviews, the USCG will also ensure the technologies are still performing in compliance with the numerical standard. In addition to these on-ship monitoring efforts, the USCG will also require vessels to report ballast water discharges and maintain records of those discharges for 24 months. All of these monitoring efforts will allow the USCG to understand the ballast water discharges and the risks associated with them and undertake adaptive management measures if problems are identified in their program.

Summary of USCG’s Ability to Prevent Harmful Exposure
The USCG is unable to prevent harmful exposure, which would have resulted in a no effect determination to listed species and their critical habitat. The USCG has minimized to the maximum extent practicable the likely adverse effects associated with this action. As noted in the approach to the assessment section, minimizing adverse effects to the maximum practicable extent may still not avoid jeopardizing listed species or adversely modifying their critical habitat because adverse effects are still expected as a result of this action.

There are two primary concerns with ballast water discharge: (1) the number of organisms per volume of water and (2) the total volume of ballast water discharged. The USCG does not have the authority to limit total volume of ballast water discharge, but they are able to limit the number of organisms per volume that can be discharged. Therefore, the USCG has selected a numerical standard that is currently the most practicably achievable. The USCG requires vessels subject to reporting requirements (identified above) to report their ballast water plans as they come in to port, prior to discharging ballast water. Mitigation measures are included in the FR to avoid environmentally sensitive areas and compliance with these mitigation measures will be reviewed by the reports as well as required record keeping on board vessels for 24 months.

Vessel operators would be required to report and maintain records of their discharges, but there is the potential for dishonesty. To make sure vessel operators are in fact treating ballast water and discharging it when they say they will, the USCG boards all vessels at least once a year determined by a decision matrix. During these exercises, the USCG determines that the vessel is in compliance with the proposed FR and has treatment technology installed. Furthermore, the USCG takes samples of the ballast water and ballast sediment to ensure the treatment technology is meeting pre-installation treatment success and being maintained to continue to meet the numerical standard. If the vessel is out of compliance, the USCG is able to take enforcement action against the vessel, as has been demonstrated previously with criminal prosecutions for falsified BWE recordkeeping and during compliance monitoring of the oil-water separator program.

In addition to reporting prior to discharge and keeping records of those discharges and treatments for 24 months, the USCG has implemented a practicability review to be carried out by 2016 that will assess whether there are new technologies available to increase the stringency of the standard. After 2016, the USCG will conduct periodic reviews every three years that will effectively be the same as the practicability review. The technology reviews will assess new technologies to determine if the numerical standard can be made more stringent. In the event of a technology that can allow vessels to practicably achieve a more stringent standard, the USCG will determine whether that technology will require an economic impact analysis. If the new technology is not economically significant, the new practicable numeric standard could be implemented in less than a year and if it is economically significant, then it would be implemented within three years.

Because the USCG was not able to avoid all possible harmful exposure associated with ballast water discharges, NMFS must assess whether and how quickly the USCG could implement new protective measures in the event that the proposed numerical standard is unable to be achieved. The USCG is going to assess the potential of all new technologies that may achieve the numerical standard under a tiered consultation process due to the quantity and variety of treatment technologies, as well as the anticipation that new technologies will continually be developed until the technologies are able to achieve more stringent standards. Essentially, once a technology is developed and can practicably increase the stringency of the ballast water standard, all vessels operators will be required to equip their vessels with that technology until a new technology is developed that can achieve that same level of stringency. Similarly, if a technology is approved but through monitoring is revealed to consistently not be able to achieve the numerical standard, the USCG would reinitiate consultation on that technology or quickly determine that particular technology is no longer approved for use and suggest one of the other technologies that is available that can meet the numerical standard. Because of all of the monitoring and reporting required by the FR, as well as the mitigation measures, the USCG is able to quickly undertake adaptive management measures if problems are identified in their program to protect listed species, their critical habitat, and the general ecosystem health within the coastal areas of the US.

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.

Government and private actions may include changes in land and water use patterns, including ownership and intensity, any of which could affect listed species or their habitat. It is difficult, and perhaps speculative, to analyze the effects of such actions, considering the broad geographic landscape covered by this BO, the geographic and political variation in the action area, extensive private land holdings, the uncertainties associated with State and local government and private actions, and ongoing changes in the region’s economy. Adverse effects to riverine habitat from continued urbanization in the basin are reasonably certain to occur. However, state and local governments have regulations in place to minimize these effects to listed species, including regulations regarding construction best management practices, storm water control, and treatment of wastewater.

Although quantifying an incremental change in survival for the species considered in this consultation due to the cumulative effects is not possible, it is reasonably likely that those effects within the action area will have a small, long-term, negative effect on the likelihood of their survival and recovery.

Conclusion

After reviewing the current status of Gulf of Maine Atlantic salmon, south Pacific loggerhead sea turtles, North Pacific loggerhead sea turtles, Hawaiian monk seals, and Cook Inlet beluga whales, the proposed action, the environmental baseline, the action area, and cumulative effects, it is NMFS’ biological opinion that the program, may affect but is not likely to adversely affect these threatened and endangered species or their critical habitat.

NMFS reviewed the current status of California coastal Chinook salmon, central valley Chinook salmon, lower Columbia River Chinook salmon, Puget Sound Chinook salmon, Snake River fall-run Chinook salmon, Snake River spring/summer-run Chinook salmon, upper Columbia River spring-run Chinook salmon, Upper Willamette River Chinook salmon, Sacramento River winter-run Chinook salmon, Hood Canal summer run chum salmon, Columbia River chum salmon, lower Columbia River coho salmon, Oregon Coast coho salmon, SONCC coho salmon, central California coast coho salmon, Snake River sockeye salmon, Ozette Lake sockeye salmon, California central valley steelhead, lower Columbia River steelhead, middle Columbia River steelhead, Puget Sound steelhead, Snake River Basin steelhead, south-central California coast steelhead, upper Columbia River steelhead, upper Willamette River steelhead, northern California steelhead, southern California steelhead, central California coast steelhead, green sturgeon, gulf sturgeon, Atlantic sturgeon, shortnose sturgeon, smalltooth sawfish, yelloweye rockfish, bocaccio, canary rockfish, Pacific eulachon, elkhorn coral, staghorn coral, white abalone, black abalone, Johnson's sea grass, Steller sea lions, spotted seals, green sea turtles, hawksbill sea turtles, loggerhead sea turtles, Kemp's ridley sea turtles, olive ridley sea turtles, leatherback sea turtles, killer whales, and humpback whales, the environmental baseline for the action area, the effects of the proposed action, and the cumulative effects. Because the program has adaptive management capabilities, feedback loops, monitoring plans, record keeping and maintaining requirements, and enforcement capabilities, it is NMFS' Opinion that the USCG ballast water management program, is not likely to jeopardize the continued existence of these endangered or threatened species.

NMFS also reviewed the current status of California coastal Chinook salmon, central valley Chinook salmon, lower Columbia River Chinook salmon, Puget Sound Chinook salmon, Snake River fall-run Chinook salmon, Snake River spring/summer-run Chinook salmon, upper Columbia River spring-run Chinook salmon, Upper Willamette River Chinook salmon, Sacramento River winter-run Chinook salmon, Hood Canal summer run chum salmon, Columbia River chum salmon, Central California coho salmon, Oregon Coast coho salmon, SONCC coho salmon, Snake River sockeye salmon, Ozette Lake sockeye salmon, California central valley steelhead, lower Columbia River steelhead, middle Columbia River steelhead, Puget Sound steelhead, Snake River Basin steelhead, south-central California coast steelhead, upper Columbia River steelhead, upper Willamette River steelhead, northern California steelhead, southern California steelhead, central California coast steelhead, Pacific eulachon, green sturgeon, gulf sturgeon, smalltooth sawfish, Steller sea lions, southern resident killer whales, green sea turtles, hawksbill sea turtles, leatherback sea turtles, loggerhead sea turtle, elkhorn coral, staghorn coral, and black abalone, the environmental baseline for the action area, the effects of the proposed action, and the cumulative effects of this program. Because the program has adaptive management capabilities, feedback loops, monitoring plans, record keeping and maintaining requirements, and enforcement capabilities, it is NMFS' Opinion that the USCG ballast water management program, is not likely to adversely modify critical habitat for these species.

INCIDENTAL TAKE STATEMENT

Section 9 of the Act and Federal regulation pursuant to section 4(d) of the ESA prohibit the take of endangered and threatened species, respectively, without special exemption. Take is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct. Harm is further defined by NMFS to include significant habitat modification or degradation that results in death or injury to listed species by significantly impairing essential behavioral patterns, including breeding, feeding, or sheltering. Harass is defined by the USFWS as intentional or negligent actions that create the likelihood of injury to listed species to such an extent as to significantly disrupt normal behavior patterns which include, but are not limited to, 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 Act provided that such taking is in compliance with the terms and conditions of this Incidental Take Statement.

Amount or Extent of Take

For this proposed action, take is authorized for each species identified in this section. However, given the complexity and limitations of scientific knowledge of the invasion process, predicting invasions, invasive species, or the expected effects is impossible at this time. Because there is no means of predicting specific invaders, the number or species of listed species that will be affected, the proportion of the population that will be taken, or the amount or extent of impacts to critical habitat, NMFS decided to assess the risks and effects of invasive species at a geographic scale using a surrogate for population and habitat scale effects.

As was explained in the exposure analysis, using the invasion rate identified using BWE, applying the anticipated reduction in propagule pressure identified in the final PEIS, and then multiplying that rate by the volume of ballast water discharged to a bioregion, it was possible to conservatively anticipate the rate of invasions to either bioregions or the US as a whole in terms of the number of new viable populations establishing each year. The volume of water in the coastal, estuarine, and freshwater regions of the US is so vast that it is not economically possible to conduct annual environmental assessments and reliably determine the number of newly established invasive species each year. It is more practicable to estimate the number of new invasions by using the best available science and measuring invasions by the volume of water discharged geographically based on the proposed numerical standard for ballast water management. Therefore NMFS will provide take in terms of the number of harmful invasive species establishing each year using volumes of ballast water discharged as a surrogate for the take that is likely to affect listed species. While the USCG does not control the volume of water discharged, they do regulate the number of organisms per volume of ballast water that is then discharged and therefore NMFS believes this is an appropriate proxy and easier to identify than the number of organisms discharged. The USCG currently requires all vessels to report all information about ballast water from uptake, treatment, and discharge, and maintain records of those reports for 24 months.

This Opinion does not reach conclusions about the risk posed by the numerical ballast water standard but rather the ballast water management program and identifies the possible, not probable, effects resulting from any non-zero numerical standard. The probable rate of species invasions is identified as is the probable rate of harmful invasions. The incidental take expected annually under this program is identified using the volume of ballast water discharged as a proxy for listed species and critical habitat affected. As the periodic reviews allow the USCG to establish more stringent ballast water standards, the amount and extent of take of listed species is expected to decrease. Assuming technology will advance in the future to allow for a new, practicable standard to be implemented, the impacts to the listed species and their critical habitat will be reduced, much as the reductions of those impacts are identified in the Opinion as the USCG moves from BWE to a numeric standard. The new levels of take resulting from the program's standard and authorized by NMFS would depend on the numerical standard implemented and the volume of ballast water being discharged by vessels, which may range between no take (no effect) and the amount of take authorized in this Opinion, but not to exceed the amount in this Opinion.

Not all newly established invasive species are harmful to listed species or their critical habitat. In fact, some may be beneficial, while most are innocuous. Past research has concluded that the proportion of invasive species that are harmful ranges from approximately 15% to 19.9%. Once an invasive species has established, co-occurring in time and space with a listed species and its critical habitat, the take that occurs is not limited to the introduction, but will persist as long as that non-native species continues to co-occur in time and space with the listed entity with harmful effects. Using the conservative figure of 19.9% of invasive species being harmful, NMFS estimates:

California coastal Chinook salmon and their critical habitat will be harmed by no more than one ballast-mediated harmful invasive species per year and no more than three in the next 16 years, which would correlate to a volumetric ballast discharge of 94×10^6 tonnes per year to Bioregion NEP-V.

Central Valley Chinook salmon and their critical habitat are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than three in the next 16 years, which would correlate to a volumetric ballast discharge of 94×10^6 tonnes per year to Bioregion NEP-V.

Lower Columbia River Chinook salmon and their critical habitat are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than one in the next 16 years, which would correlate to a volumetric ballast discharge of 94×10^6 per year tonnes to Bioregion NEP-IV.

Snake River fall-run Chinook salmon and their critical habitat are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than one in the next 16 years, which would correlate to a volumetric ballast discharge of 94×10^6 tonnes per year to Bioregion NEP-IV.

Snake River spring/summer-run Chinook salmon and their critical habitat are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than one in the next 16 years, which would correlate to a volumetric ballast discharge of 94×10^6 tonnes per year to Bioregion NEP-IV.

Upper Columbia River Chinook salmon and their critical habitat are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than one in the next 16 years, which would correlate to a volumetric ballast discharge of $94x10^6$ tonnes per year to Bioregion NEP-IV.

Upper Willamette River Chinook salmon and their critical habitat are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than one in the next 16 years, which would correlate to a volumetric ballast discharge of $94x10^6$ tonnes per year to Bioregion NEP-IV.

Puget Sound Chinook salmon and their critical habitat are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than one in the next 16 years, which would correlate to a volumetric ballast discharge of $94x10^6$ tonnes to Bioregion NEP-IV.

Sacramento winter-run Chinook salmon and their critical habitat are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than three in the next 16 years, which would correlate to a volumetric ballast discharge of $94x10^6$ tonnes per year to Bioregion NEP-V.

Columbia River chum salmon and their critical habitat are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than one in the next 16 years, which would correlate to a volumetric ballast discharge of $94x10^6$ tonnes per year to Bioregion NEP-IV.

Hood Canal summer-run chum salmon and their critical habitat are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than one in the next 16 years, which would correlate to a volumetric ballast discharge of $94x10^6$ tonnes per year to Bioregion NEP-IV.

Lower Columbia River coho salmon are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than one in the next 16 years, which would correlate to a volumetric ballast discharge of $94x10^6$ tonnes per year to Bioregion NEP-IV.

SONCC coho salmon and their critical habitat are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than three in the next 16 years, which would correlate to a volumetric ballast discharge of $94x10^6$ tonnes per year to Bioregions NEP-IV and NEP-V.

Central California coast coho salmon and their critical habitat are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than three in the next 16 years, which would correlate to a volumetric ballast discharge of $94x10^6$ tonnes per year to Bioregion NEP-V.

Oregon Coast coho salmon and their critical habitat are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than one in the next 16 years, which would correlate to a volumetric ballast discharge of 94×10^6 tonnes per year to Bioregion NEP-IV.

Snake River sockeye salmon and their critical habitat are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than one in the next 16 years, which would correlate to a volumetric ballast discharge of 94×10^6 tonnes per year to Bioregion NEP-IV.

Ozette Lake sockeye salmon and their critical habitat are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than one in the next 16 years, which would correlate to a volumetric ballast discharge of 94×10^6 tonnes per year to Bioregion NEP-IV.

California central valley steelhead and their critical habitat are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than three in the next 16 years, which would correlate to a volumetric ballast discharge of 94×10^6 tonnes per year to Bioregion NEP-V.

Lower Columbia River steelhead and their critical habitat are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than one in the next 16 years, which would correlate to a volumetric ballast discharge of 94×10^6 tonnes per year to Bioregion NEP-IV.

Middle Columbia River steelhead and their critical habitat are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than one in the next 16 years, which would correlate to a volumetric ballast discharge of 94×10^6 tonnes per year to Bioregion NEP-IV.

Northern California steelhead and their critical habitat are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than three in the next 16 years, which would correlate to a volumetric ballast discharge of 94×10^6 tonnes per year to Bioregion NEP-V.

Snake River Basin steelhead and their critical habitat are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than one in the next 16 years, which would correlate to a volumetric ballast discharge of 94×10^6 tonnes per year to Bioregion NEP-IV.

South-central California coast steelhead and their critical habitat are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than three in the next 16 years, which would correlate to a volumetric ballast discharge of 94×10^6 tonnes per year to Bioregion NEP-V.

Southern California steelhead and their critical habitat are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than one in the next 16 years, which would correlate to a volumetric ballast discharge of $94x10^6$ tonnes per year to Bioregion NEP-IV.

Upper Columbia River steelhead and their critical habitat are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than one in the next 16 years, which would correlate to a volumetric ballast discharge of $94x10^6$ tonnes per year to Bioregion NEP-IV.

Upper Willamette River steelhead and their critical habitat are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than one in the next 16 years, which would correlate to a volumetric ballast discharge of $94x10^6$ tonnes per year to Bioregion NEP-IV.

Puget Sound steelhead are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than one in the next 16 years, which would correlate to a volumetric ballast discharge of $94x10^6$ tonnes per year to Bioregion NEP-IV.

Central California coast steelhead and their critical habitat are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than three in the next 16 years, which would correlate to a volumetric ballast discharge of $94x10^6$ tonnes per year to Bioregion NEP-V.

Pacific eulachon and their critical habitat are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than three in the next 16 years, which would correlate to a volumetric ballast discharge of $94x10^6$ tonnes per year to Bioregions NEP-IV and NEP-V.

Shortnose sturgeon are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than eight in the next 16 years, which would correlate to a volumetric ballast discharge of $94x10^6$ tonnes per year to Bioregions NA-ET2, NA-ET3, and CAR-VII.

Gulf of Maine DPS Atlantic sturgeon are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than eight in the next 16 years, which would correlate to a volumetric ballast discharge of $94x10^6$ tonnes per year to Bioregions NA-ET2, NA-ET3, and CAR-VII.

New York Bight DPS Atlantic sturgeon are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than eight in the next 16 years, which would correlate to a volumetric ballast discharge of $94x10^6$ tonnes per year to Bioregions NA-ET2, NA-ET3, and CAR-VII.

Chesapeake Bay DPS Atlantic sturgeon are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than eight in the next 16 years, which would correlate to a volumetric ballast discharge of $94x10^6$ tonnes per year to Bioregions NA-ET2, NA-ET3, and CAR-VII.

Carolina DPS Atlantic sturgeon are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than eight in the next 16 years, which would correlate to a volumetric ballast discharge of 94×10^6 tonnes per year to Bioregions NA-ET2, NA-ET3, and CAR-VII.

South Atlantic DPS Atlantic sturgeon are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than eight in the next 16 years, which would correlate to a volumetric ballast discharge of 94×10^6 tonnes per year to Bioregions NA-ET2, NA-ET3, and CAR-VII.

Green sturgeon and their critical habitat are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than three in the next 16 years, which would correlate to a volumetric ballast discharge of 94×10^6 tonnes per year to Bioregion NEP-V.

Gulf sturgeon and their critical habitat are likely to be harmed by no more than two ballast-mediated harmful invasive species per year and no more than 16 in the next 16 years, which would correlate to a volumetric ballast discharge of 188×10^6 tonnes per year to Bioregion CAR-I.

Yelloweye rockfish are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than one in the next 16 years, which would correlate to a volumetric ballast discharge of 94×10^6 tonnes per year to Bioregion NEP-IV.

Bocaccio are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than one in the next 16 years, which would correlate to a volumetric ballast discharge of 94×10^6 tonnes per year to Bioregion NEP-IV.

Canary rockfish are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than one in the next 16 years, which would correlate to a volumetric ballast discharge of 94×10^6 tonnes per year to Bioregion NEP-IV.

Smalltooth sawfish and their critical habitat are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than five in the next 16 years, which would correlate to a volumetric ballast discharge of 94×10^6 tonnes per year to Bioregion CAR-IV and the Florida ports of CAR-I.

Elkhorn coral and their critical habitat are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than five in the next 16 years, which would correlate to a volumetric ballast discharge of 94×10^6 tonnes per year to Bioregion CAR-IV and the Florida ports of CAR-I.

Staghorn coral and their critical habitat are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than five in the next 16 years, which would correlate to a volumetric ballast discharge of 94×10^6 tonnes per year to Bioregion CAR-IV and the Florida ports of CAR-I.

White abalone are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than three in the next 16 years, which would correlate to a volumetric ballast discharge of 94×10^6 tonnes per year to Bioregion NEP-V.

Black abalone and their critical habitat are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than three in the next 16 years, which would correlate to a volumetric ballast discharge of 94×10^6 tonnes per year to Bioregion NEP-V.

Eastern DPS Steller sea lions and their critical habitat are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than five in the next 16 years, which would correlate to a volumetric ballast discharge of 94×10^6 tonnes per year to Bioregions NEP-IV and NEP-V.

Western DPS Steller sea lions and their critical habitat are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than four in the next 16 years, which would correlate to a volumetric ballast discharge of 94×10^6 tonnes per year to Bioregions NEP-I and NEP-II.

Spotted seals are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than four in the next 16 years, which would correlate to a volumetric ballast discharge of 94×10^6 tonnes per year to Bioregions NEP-I, NEP-II, and NA-S1.

Southern resident killer whales and their critical habitat are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than one in the next 16 years, which would correlate to a volumetric ballast discharge of 94×10^6 tonnes per year to Bioregion NEP-IV.

Humpback whales are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than 13 in the next 16 years, which would correlate to a volumetric ballast discharge of 94×10^6 tonnes per year to Bioregions SP-XXI, IX, XII, NEP-I, NEP-II, NEP-V, NEP-VI, NEP-IV, NA-ET3, NA-ET2, and CAR-VII.

Green sea turtles and their critical habitat are likely to be harmed by no more than two ballast-mediated harmful invasive species per year and no more than 23 in the next 16 years, which would correlate to a volumetric ballast discharge of 188×10^6 tonnes per year to Bioregions SP-XXI, SP-IX, SP-XII, NA-ET3, CAR-VII, CAR-IV, CAR-I, and NEP-VI.

Leatherback sea turtles and their critical habitat are likely to be harmed by no more than two ballast-mediated harmful invasive species per year and no more than 24 in the next 16 years, which would correlate to a volumetric ballast discharge of 188×10^6 tonnes per year to Bioregions SP-XXI, SP-IX, SP-XII, NA-ET3, CAR-VII, CAR-IV, CAR-I, NEP-VI, and NEP-V.

Kemp’s Ridley sea turtles are likely to be harmed by no more than two ballast-mediated harmful invasive species per year and no more than 23 in the next 16 years, which would correlate to a volumetric ballast discharge of 188x10^6 tonnes per year to Bioregions NA-ET3, CAR-VII, CAR-IV, and CAR-I.

Olive ridley sea turtles are likely to be harmed by no more than two ballast-mediated harmful invasive species per year and no more than 18 in the next 16 years, which would correlate to a volumetric ballast discharge of 188x10^6 tonnes per year to Bioregions SP-XXI, SP-IX, SP-XII, CAR-I, and NEP-VI.

North Pacific loggerhead sea turtles are likely to be harmed by no more than one ballast-mediated harmful invasive species per year and no more than five in the next 16 years, which would correlate to a volumetric ballast discharge of 94x10^6 tonnes per year to Bioregions SP-XXI, NEP-VI, and NEP-V.

North Atlantic loggerhead sea turtles are likely to be harmed by no more than two ballast-mediated harmful invasive species per year and no more than 23 in the next 16 years, which would correlate to a volumetric ballast discharge of 188x10^6 tonnes per year to Bioregions NA-ET3, CAR-VII, CAR-IV, and CAR-I.

Hawksbill sea turtles and their critical habitat are likely to be harmed by no more than two ballast-mediated harmful invasive species per year and no more than 23 in the next 16 years, which would correlate to a volumetric ballast discharge of 188x10^6 tonnes per year to Bioregions SP-XXI, SP-IX, SP-XII, NA-ET3, CAR-VII, CAR-IV, and CAR-I.

This Opinion calculates the likely number of invasive species introduced using the volume of ballast water discharged as a surrogate, then provides take coverage for harmful invasions by calculating the ratio of harmful invasive species identified from several hundred previous invasive species that have been studied. Under a new, more stringent ballast water standard, the number of non-native species per volume of ballast water would be reduced. This will reduce the probability of a harmful invasive species establishing, and therefore require a new proxy for incidental take. However, as long as establishing a new more stringent numerical standard would operate within the requirements of the ballast water management program as were assessed in this Opinion, then identifying a more stringent numerical standard would not require reinitiation of consultation, but rather it would require NMFS to amend this Opinion to identify the new volume of ballast water that would act as a surrogate for take under a new practicable ballast water standard. NMFS would also be required to provide the amended Opinion with a new incidental take statement to the USCG.

Reasonable and Prudent Measures

Reasonable and Prudent Measures (RPMs) are non-discretionary measures to minimize take that may or may not already be part of the description of the proposed action. They must be implemented as binding conditions for the exemption in section 7(o)(2) to apply. The USCG has the continuing duty to regulate the activities covered in this incidental take statement. If the USCG fails to adhere to the terms and conditions of the incidental take statement through enforceable terms that are added to this document, or fails to retain the oversight to ensure compliance with these terms and conditions, the protective coverage of section 7(o)(2) may lapse. Activities which do not comply with all relevant RPMs will require further consultation.

NMFS believes that the following RPM is necessary and appropriate to minimize take of listed species and their critical habitat resulting from implementation of this action.

The USCG shall monitor and report to NMFS Office of Protected Resources various components of the ballast water management program.

Terms and Conditions

To be exempt from the prohibitions of section 9 of the ESA, the action must be implemented in compliance with the following terms and conditions, which implement the RPM described above. These terms and conditions are non-discretionary.

To implement the RPM (monitoring and reporting):

1. Provide the NBIC report summarizing national patterns of ballast water delivery and management.

2. Provide results of the practicability review.

3. Provide results of any periodic programmatic reviews which may be undertaken as mandated under NISA/NANPCA.

4. Provide an annual report detailing the number of ships entering US waters, the number of inspections, ballast water violations found, and corrective actions taken to fix the violation.

CONSERVATION RECOMMENDATIONS

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

1. *Address Other Sources of Invasive Species*. The USCG should establish mandatory requirements of vessels that will, along with ballast water treatment, more effectively reduce the potential of invasive species. These requirements should focus on hull fouling, sea chests, anchor chains, and any other source for a vessel to bring non-native species into waters of the US.

2. *Incentives*. The USCG, to best implement the goals of NISA and the ESA, should establish economic incentives or grant programs with the goal of creating more stringent ballast water standards by developing new technologies and testing procedures.

3. *Habitat Improvement Measures*. The USCG should implement habitat restoration measures in areas where non-native species have established and are harmful to listed species or their critical habitat. The goal of these measures should be to directly counter the adverse effects of the non-native species to mitigate the impacts to listed species and their critical habitat.

4. *Alternative Discharge Options*. The USCG should work with US port owners to create on-site (land based or vessel-based) discharge facilities capable of receiving ballast water discharges from vessels at the port. The USCG should consider funding these facilities under 7(a)(1) of the ESA in an effort to eventually eliminate ballast water discharges and invasive species introductions to waters of the US.

5. *Eliminate Discharges*. Several vessels were identified in the final PEIS that could operate without discharging ballast water. The USCG should consider making it mandatory that the vessels that can operate without discharging ballast water always operate without discharging ballast water except when holding ballast water onboard would create an unsafe situation.

6. *Collaboration with EPA*. The USCG should continue working with EPA to identify jurisdictional boundaries that can be unclear between the NISA and CWA, both within the federal and state context, so as to maximize compliance under NISA. In particular, a major pathway of new invasive species is from hull fouling, yet neither agency is currently regulating the discharges of those non-native propagules.

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

REINITIATION NOTICE

This concludes formal consultation on the USCG’s National Ballast Water Management Program. 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 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 the event USCG would authorize, fund, or carry out ballast water discharge related activities that may affect resources in a manner or to an extent not considered in this Opinion, USCG must reinitiate consultation to compensate for information that was not available for consideration during this consultation.

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