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Endangered Species Act Section 7 Consultation
Biological Opinion & Conference Opinion
On the
U.S. Environmental Protection Agency's
Approval of State or Tribal, or Federal Numeric Water Quality Standards for Cyanide
Based on EPA's Recommended 304(a) Aquatic Life Criteria

National Marine Fisheries Service
Office of Protected Resources
Silver Spring, MD 20910

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Agency: U.S. Environmental Protection Agency

Activities Considered: Approval of State or Tribal, or Federal Numeric Water Quality Standards for Cyanide Based on EPA's Recommended 304(a) Aquatic Life Criteria

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

Approved by: ____

Date: ____

Section 7(a)(2) of the Endangered Species Act of 1973, as amended (ESA; 16 U.S.C. 1539(a)(2)) 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. When a federal agency's action "may affect" a protected species, that agency is required to consult formally with the National Marine Fisheries Service (NMFS) or the U.S. Fish and Wildlife Service (FWS; together, the Services), depending upon the endangered species, threatened species, or designated critical habitat that may be affected by the action (50 CFR 402.14(a)). Federal agencies are exempt from this general requirement if they have concluded that an action "may affect, but is not likely to adversely affect" endangered species, threatened species, or designated critical habitat and NMFS or the FWS concur with that conclusion (50 CFR 402.14(b)).

The U.S. Environmental Protection Agency (EPA) initiated formal consultation with NMFS and the FWS on their recommended 304(a) criteria and the approval of state and tribal water quality standards, or federal water quality standards promulgated by EPA that are identical to or more stringent than the section 304(a) aquatic life criteria published pursuant to the Clean Water Act (CWA; 33 U.S.C. 1251 *et seq*.), for the protection of aquatic life from harmful effects of cyanide (CN). This document represents NMFS' biological and conference opinion (Opinion) on EPA's approval of numeric standards for cyanide in fresh and salt waters of the U.S and its effects on threatened and endangered species, their designated critical habitat, and species proposed for listing as threatened or endangered, and critical habitat proposed for designation. This consultation does not address the effects of specific modifications of these criteria that are undertaken by states and tribes or the permits issued by particular states or tribes. This Opinion contains a detailed explanation of the particular circumstances warranting subsequent consultation (tiered consultations) with NMFS' Regional Offices in the section titled *Application* *of this Consultation to Other EPA Actions*.

This Opinion is based on our review of the EPA's *Biological Evaluation of Aquatic Life Criteria- Cyanide*, status reviews, listing documents, and recovery plans for the threatened and endangered species under NMFS' jurisdiction, reports on the status and trends of water quality in the United States that have been prepared by the U.S. Geological Survey, EPA, and others, past and current research and population dynamics modeling efforts, and published and unpublished scientific information on the biology and ecology of threatened and endangered sea turtles, marine mammals, salmon, sturgeon, sawfish, abalone, and seagrasses in the action area, and other sources of information which are discussed in greater detail in the *Approach to the Assessment* section of this Opinion.. This Opinion has been prepared in accordance with section 7 of the ESA and associated implementing regulations.

Consultation History

On January 18, 2001, the Services and EPA signed a Final Memorandum of Agreement (MOA) on the enhanced coordination under the ESA and the CWA. The final MOA published in the *Federal Register* on February 22, 2001 (66 FR 36) and described, among other things, a plan for assisting EPA in meeting it's section 7 responsibilities on two CWA programs: water quality standards and the National Pollutant Discharge Elimination System (NPDES) permits program.

In January 2004, the Services and EPA decided to proceed with a data call for the first batch of pollutants that would be reviewed in consultation, while continuing to work on the *Draft Methodology for Conducting Biological Evaluations of Aquatic Life Criteria--Methods Manual.*

On May 14, 2004, the Services and EPA issued data calls to the regional staff and science center staff requesting information and data on cyanide, ammonia, chromium III and chromium VI. The data call requested regions and science centers send relevant studies to Headquarters by June 30, 2004.

On November 12, 2004, EPA provided the Services a revised *Draft Methodology for Conducting Biological Evaluations of Aquatic Life Criteria--Methods Manual* (dated October 29, 2004, on the document). This version represented a methodology developed collaboratively, and which had been peer reviewed by subject experts outside of the Federal government.

In December 2004, NMFS and EPA exchanged comments on recommended revisions to the November draft methodology. EPA also informed the Services that they had received a draft BE for cyanide from their contractor and were reviewing the document to ensure the contractor had followed the BE methodology accurately.

On January 24, 2005, EPA emailed NMFS a partial draft of their CN BE.

On May 3, 2005, the Services jointly provided comments to EPA on their January 19, 2005, draft biological evaluation for cyanide criteria.

On January 26, 2006, EPA provided NMFS with a draft CN BE and requested a review of the

BE's "completeness" in fulfilling the information requirements for section 7 consultation. On April 21, 2006, NMFS provided comments to EPA on the "completeness" of the draft BE.

In a June 29, 2006, letter, EPA requested NMFS' concurrence with their determination that proposed approval of cyanide criteria was not likely to adversely affect all listed species and critical habitat under NMFS' jurisdiction.

On November 11, 2006, the FWS sent NMFS a copy of EPA's revised *Draft Framework for Conducting Biological Evaluations of Aquatic Life Criteria: Methods Manual*, which EPA revised and submitted to FWS in July 31, 2006 and which EPA used to support their effects determinations.

On November 15, 2006, NMFS sent EPA a letter with a detailed explanation as to why NMFS could not concur with EPA's determinations that the recommended water quality standards for cyanide "may affect, but are not likely to adversely affect" threatened and endangered species and designated critical habitat.

On March 23, 2007, EPA requested formal consultation supported by their March 23, 2007, *Biological Evaluation of Aquatic Life Criteria—Cyanide*, which concluded their action was "not likely to jeopardize the continued existence of any federally listed species or result in the destruction or adverse modification of designated critical habitat [*sic*]."

On June 21, 2007, NMFS sent a letter to EPA acknowledging the initiation of formal consultation. NMFS' letter acknowledged that the scope and complexity of the national consultation on the aquatic life criteria for cyanide may require more time than usual to complete the biological opinion.

On May 5-9, 2008, the Services met with EPA to conduct a "Kaizen" "lean event." The purpose of the meetings was to analyze the cyanide consultation process from the development of a biological assessment through the anticipated completion of formal consultation in an effort to find efficiencies in the process. The Services and EPA also discussed coordination and communication with respect to the national consultation on cyanide and local consultation on EPA promulgation of Oregon water quality standards.

On June 12, 2008, the Services and EPA met to follow up on the Kaizen lean event. Subsequent follow up meetings were cancelled until the Services completed draft biological opinions.

BIOLOGICAL & CONFERENCE OPINION

Description of the Proposed Action

The action considered in this Opinion, and beginning a series of national water quality consultations with EPA, is EPA's continuing approval of state or tribal water quality standards, or federal water quality standards promulgated by EPA, that are identical to or more stringent than EPA's recommended 304(a) aquatic life criteria for cyanide. These water quality standards define water column concentrations of cyanide that should protect against adverse ecological effects to aquatic life in fresh and salt water. The 304(a) aquatic life criteria recommendations, which are the foundation for many approved 303(c) standards, are designed to protect aquatic organisms from unacceptable toxicity during acute (short) and chronic (long) exposures in the water column. The intent is to define a level in the waterbody of a pollutant that will be fully protective of the designated use and which a regulatory authority may use in adopting regulatory water quality standards and thereby control, reduce, or eliminate discharges of that pollutant (BE page 11).

Section 304(a)(1) of the CWA directs EPA to publish criteria for water quality accurately reflecting the latest scientific knowledge on a number of factors including "… the kind and extent of all identifiable effects on health and welfare including, but not limited to, plankton, fish, shellfish, wildlife, plant life, shorelines, beaches, esthetics, and recreation which may be expected from the presence of pollutants in any body of water, including ground water; on the concentration and dispersal of pollutants, or their byproducts, through biological, physical and chemical processes; and on the effects of pollutants on biological community diversity, productivity, and stability including information on the factors affecting rates of eutrophication and rates of organic and inorganic sedimentation for varying types of receiving waters." The water quality standards program is authorized under section 303(c) of the CWA (33 U.S.C. 1313(c)) and directs states to adopt numeric criteria for specific toxic pollutants that appear on a priority pollutant list[1] and for which EPA published 304(a) criteria recommendations. States can, pursuant to section 303(c) of the CWA, adopt water quality standards that differ from EPA's 304(a) criteria values whenever adequately justified, but states and tribes generally choose to adopt EPA's 304(a) criteria verbatim. Once adopted into state water quality standards, criteria form the legal basis for implementing the CWA programs to control pollution and achieve the goals and requirements of the CWA.

The purpose of these national consultations is to assess the effect of the EPA's 304(a) criteria recommendation and the subsequent approval of state and tribal water quality standards, or federal water quality standards promulgated by EPA that are identical to or more stringent than the section 304(a) aquatic life criteria on threatened and endangered species and their designated critical habitat (together, listed resources), and species and critical habitat that are proposed for

[1] Section 307(a) of the CWA, which defines priority pollutants as compounds and families that are among the most persistent, prevalent and toxic chemicals.

listing or designation (together, proposed resources). In particular, this Opinion analyzes whether EPA's approval of state standards that rely on the national criteria for cyanide are not likely to jeopardize the continued existence of threatened and endangered species (including species proposed for listing as threatened or endangered), or result in the destruction or adverse modification of designated critical habitat (see the BE, page 1).

In 1985 EPA published two values for cyanide pollution in each fresh and salt "waters of the United States," the criterion maximum concentration (CMC) and the criterion continuous concentration (CCC). EPA's ambient water quality criteria for cyanide are expressed as free cyanide (Table 1). The CMC represents EPA's estimate of the highest concentration of cyanide in fresh or salt water to which an aquatic community's brief exposure (acute limit) would not result in an unacceptable effect. The CMC is derived from a set of LC50 values for a variety of aquatic species. The LC50 value is the lethal concentration of a chemical that causes 50% mortality, immobilization, or loss of equilibrium in the test organism in 48 to 96-hour laboratory tests. The CMC is then set to one-half of the fifth percentile of the genus mean acute value (GMAV) for the various species tested to provide a level of protection that is better than 50% mortality. EPA recommends that the one-hour average exposure concentrations should not exceed the CMC more than once every three years on the average, making such exceedances a relatively rare event (EPA 1991).

Table 1. Cyanide 304(a) Aquatic Life Criteria (in μg/L of free cyanide [EPA 1985])

Medium	Criterion Maximum Concentration	Criterion Continuous Concentration
Fresh water	22.36	5.221
Saltwater	1.015	1.015

The CCC represents EPA's estimate of the highest concentration of cyanide in either fresh or salt water, to which an aquatic community's prolonged exposure (chronic limit) would not result in an unacceptable effect. The CCC is derived from a set of chronic values, which are the geometric mean of the highest no observed effect concentrations (NOECs) and lowest observed effect concentrations (LOECs) for survival, growth, or reproduction in tests that range from seven days to several months. EPA sets the CCC to the estimated fifth percentile of the chronic values either by direct calculation or by using the acute-to-chronic ratios (ACRs). For the CCC, EPA recommends that the four-day average exposure concentrations should not exceed the CCC more frequently than once every three years on average (EPA 1991).

Approach to the Assessment

Section 7(a)(2) of the ESA of 1973, as amended (16 U.S.C. 1536(a)(2)), requires federal agencies, in consultation with and with the assistance of the Services, to ensure that any action they authorize, fund, or carry out is not likely to jeopardize the continued existence of endangered species or threatened species or result in the destruction or adverse modification of designated critical habitat. When NMFS consults with federal agencies to help them comply with this requirement, we first assess the direct and indirect effects of the proposed federal action to determine whether the proposal is likely to (a) appreciably increase a species' extinction probability (or reduce their probability of being conserved or recovered) or (b) appreciably reduce the conservation value of critical habitat that has been designated for one or more of those species. If we conclude that one of these outcomes is likely, we work with the federal agency, applicant, or both, to develop alternatives that avoid this likelihood.

NMFS approaches its section 7 analyses through a series of steps. The first step identifies those aspects of proposed actions that are likely to have individual, interactive, or cumulative direct and indirect effects on the environment (the potential stressors of an action). As part of this step, we identify the spatial extent of these stressors, including changes in their spatial extent over time. The spatial extent of these stressors represents the *Action Area* for consultation.

To begin the second step of our analyses, we determine whether endangered species, threatened species or designated critical habitat are likely to occur in the same space and the same time as the potential stressors. These species then become the focus of our *Exposure Analysis*. As our point of reference for evaluating the risk posed by their exposure, we rely on our understanding of the condition of the species and the conservation value of critical habitat, and any biological and ecological information on the species and their critical habitat that is relevant to our effects analysis (this information is represented in the *Status of the Species and Critical Habitat*). In the status of the species section of our Opinion, we review the species' legal status, trends, and the threats that led to this status as well as those that may be impeding the species' chances of recovery. Our assessment is also informed by the effects of past and ongoing human and natural factors leading to the current status of the species, its habitat, and ecosystem. This information is presented in the *Environmental Baseline*. By regulation, the environmental baseline for an action includes the past and present impacts of all federal, state, or private actions and other human activities in an action area, and 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. The environmental baseline is designed to assess the condition of the habitat and the species within the action area.

Often, NMFS will combine the status of the species and the environmental baseline where the status encompasses the entire range of a species. In this Opinion, we address the two separately, focusing the environmental baseline on the current condition of the nation's fresh water and marine aquatic habitats. In some cases we address watersheds that may not contain listed species under NMFS' jurisdiction because the watershed influences coastal conditions where listed marine and anadromous species occur. Our summary of the environmental baseline complements the information provided in the status of the species section of this Opinion, and provides information on the past and present ecological conditions of the action area that is necessary to further understand the species' current risk of extinction.

Our effects analyses, summarized in the *Effects of the Action* section of this Opinion, identify the nature of the listed species and critical habitat co-occurrence with the effects of the action over space and time (their exposure). Our exposure analyses identify the physical or biological features of critical habitat that would be exposed to the action, including any listed primary constituent elements that require special management consideration or protection such as sites for breeding and rearing, food, water, space for growth and normal behavior, and cover and shelter; and we identify the number, age or life stage, and gender of the individuals that are likely to be exposed to an action's effects and the populations or subpopulations those individuals represent. Once we identify the individuals and populations, or constituent laments that are likely to be exposed to an action's effects and the nature of that exposure, we examine the scientific and commercial data available to determine whether and how those listed species and their critical habitat (collectively termed listed resources) are likely to respond given their exposure (these represent our *response analyses*). The final steps of our analyses—establishing the risks those responses pose to listed resources—are different for listed species and designated critical habitat (these represent our *risk analyses*).

Our jeopardy determinations must be based on an action's effects on the continued existence of threatened or endangered species as those "species" have been listed, which can include 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 species depends on the viability of the populations that comprise them. Similarly, the continued existence of populations are determined by the fate of the individuals that comprise them; populations grow or decline as the individuals that comprise the population live, die, grow, mature, migrate, and reproduce (or fail to do so). Our risk analyses reflect the relationships between the listed species and the populations that comprise them, and the individuals that comprise those populations. Our risk analyses begin by identifying the probable risks actions pose to listed individuals that are likely to be exposed to an action's effects. Our analyses then integrate those individuals' risks to identify consequences to the populations they represent and next we determine the consequences of population-level effects on the species as listed.

We measure risks to listed individuals using the individual's "fitness," which are changes in an individual's growth, survival, annual reproductive success, or lifetime reproductive success. In particular, we examine the scientific and commercial data available to determine if an individual's probable responses to an action's effect on the environment (which we identify during our response analyses) are likely to have consequences for the individual's fitness. When individual listed plants or animals are expected to experience reductions in fitness, we would expect those reductions to also reduce the abundance, reproduction rates, or growth rates (or increase variance in one or more of these rates) of the populations those individuals represent (see Stearns 1992). A reduction 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 itself is a *necessary* condition for reductions in a species' viability. On the other hand, when listed plants or animals exposed to an action's effects are not expected to experience reductions in fitness, we would not expect the action to have adverse consequences on the viability of the populations those individuals represent or the species those populations comprise (for example, see Anderson 2000, Mills and Beatty 1979, Stearns 1992). If we conclude that listed plants or animals are not likely to experience reductions in their fitness we would conclude our assessment.

If, however, we conclude that listed plants or animals are likely to experience reductions in their fitness, our assessment examines if those reductions are likely to be sufficient to reduce the viability of the populations those individuals represent (measured using changes in the population's abundance, reproduction, spatial structure and connectivity, growth rates, genetic health, or variance in these measures to make inferences about the population's extinction risks). In this step of our analyses, we use the population's base condition (established in the *Environmental Baseline* and *Status of Listed Resources* sections of this Opinion) as our point of reference.

Our assessment framework assumes—an assumption that is supported by published evidence—that the health and fitness of individual plants or animals will integrate the effects of the physical, chemical, and biological phenomena they are exposed to during their lifetimes and at specific developmental stages of their lives. That is, our assessment framework assumes that the total effects of exposing an animal to a suite of stressors, for example, coho salmon to a combination of toxic chemicals and an altered hydrograph from various flow controls will appear as a reduction in the fitness (reductions in annual or lifetime reproductive success) of individual coho salmon thus exposed. If exposing endangered or threatened marine and anadromous animals to chemical pollutants interacts with their exposure to other anthropogenic stressors, such as construction noise or disturbance or other toxic chemicals, and produces consequences that would not occur without that interaction, the consequence would appear as a reduction in performance of the individual animals.

Thus our assessment of the impact of the proposed action begins by considering the impact of the environmental baseline on the fitness of the individuals in the action area. As part of this assessment, we must consider how listed individuals are likely to respond to any interactions and synergisms between the proposed action and its stressors, pre-existing stressors and experience (represented by the *Status of the Species* and *Environmental Baseline*, as well as those stressors that are reasonably certain to occur in the action area for the future life of the action (represented by *Cumulative Effects*). If we conclude that listed individuals are likely to experience reductions in their annual or lifetime reproductive success, we then ask if those reductions are likely to be sufficient to reduce the viability of the populations those individuals represent (measured using changes in the population's abundance, reproduction, spatial structure and connectivity, genetic health, growth rates, or variance in these measures to make inferences about the population's probability of becoming extinct). Finally, if we conclude that the viability of one or more populations of a listed species is likely to be reduced, we determine whether that reduction is likely to be sufficient to reduce the viability of the species those populations comprise (here, a species' "viability" is its probability of becoming extinct or of being "recovered" to the point at which the protections of the ESA are no longer necessary or warranted). In this step of our analyses, we use the species' status as our point of reference.

For designated critical habitat, our destruction or adverse modification determinations must be based on an action's effects on the conservation value of habitat that has been designated as critical.[2] If an area encompassed in a critical habitat designation is likely to be exposed to the direct or indirect consequences of the proposed action on the natural environment, we ask if

[2] Several courts have ruled the definition of destruction or adverse modification that appears in the section 7 regulations at 50 CFR 402.02 as invalid. Consequently, we do not rely on that definition for the determinations we make in this Opinion. Instead, we use the conservation value of critical habitat for our determinations which focuses on the designated area's ability to contribute to the conservation of the species for which the area was designated.

constituent elements included in the designation (if there are any) or physical, chemical, or biotic phenomena that give the designated area value for the conservation of the species, are likely to respond to that exposure. If those constituent elements (or phenomena) are likely to respond, we ask if those responses are likely to be sufficient to reduce the quantity, quality, or availability of those constituent elements or physical, chemical, or biotic phenomena. If the conservation value is reduced, we then ask if those reductions are likely to be sufficient to reduce the conservation value of the entire critical habitat designation.

National Programmatic Consultations

Our national programmatic consultations typically analyze the general environmental consequences of a broad scope of actions or policy alternatives under consideration by a federal agency. In these types of consultations we focus on the general patterns associated with an agency's decision to authorize a particular national or programmatic action. Subsequent consultations that "tier" off of these national consultations, when warranted, would analyze the project and site specific effects typical of most consultations. Any subsequent section 7 consultations conducted by NMFS personnel would be designed to determine whether and to what degree the specific action under review fits within the general pattern identified in the national consultations, and would 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.

Thus, our national programmatic consultations focus on the evidence available to determine whether and to what degree the agency's action is likely to prevent exposure, or mitigate the responses or risks any responses would pose to listed species or their designated critical habitat. An agency can generally satisfy this requirement when the action contains features that: (1) prevent listed resources from being exposed to subsequent actions or their direct or indirect effects; (2) mitigate how listed resources respond to that exposure, when listed resources are exposed to the actions and their effect; or (3) mitigate the risks any responses pose to listed individuals, populations, species, or designated critical habitat when listed resources are likely to be exposed and respond to that exposure.

In examining an agency's program, we would examine the general activities the agency would authorize, fund or carry out. The steps of the national-level assessment remain much the same as described for our site-specific consultation, as outlined earlier in this section. National broad scale assessments and programmatic assessments, however, are necessarily focused on whether and to what degree a federal action can ensure that actions taken under the program are not likely to individually or cumulatively, jeopardize the continued existence of endangered or threatened species and are not likely to result in the destruction or adverse modification of designated critical habitat. Our description of the probable responses of the listed resources to the national action and the risks the national action poses to those listed resources is at the core of our evaluation, and is informed by the general patterns we observed through prior experience with an agency's actions or classes of activities.

The conceptual model NMFS uses for national consultations focuses on four main elements of action agency's national action: (1) the decision-making process an action agency uses to authorize, fund, or carry out national actions; (2) the national action, and any subsequent actions or activities the agency would authorize, fund or carry out in accordance with the national action; (3) the intended and unintended consequences that are likely to result from authorized activities; and (4) the mechanisms that improve the agency's action(s) over time. We begin our national 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 (or recommend actions) that may (or may not) require specific actions undergo subsequent review and decision-making.

An agency's decision-making process will normally identify certain standards that an action must satisfy before an agency would authorize, fund or carry them out. Generally, decision-making involves hard or formal procedures (such as agency regulatory procedures and public noticing requirements), soft or flexible information standards (e.g., agency "guidelines", and the best professional judgments personnel make when considering conflicting information and making recommendation in the face of uncertainty). These procedures outline how the agency would decide whether or not to authorize, fund or carry out specific actions. Typically an agency's decision making process is shaped to respond to:

- the statutory and regulatory standards an action must satisfy before the agency would authorize, fund, or carry them out;
- any data and other information the agency must gather and evaluate to satisfy their statutory and regulatory requirements, as well as requirements of the Administrative Procedure Act, Information Quality and related administrative statutes, like the Paperwork Reduction Act, Regulatory Flexibility Act, and so on.
- the agency's obligation to review and analyze the relevant information within the context of applicable standards to ensure that specific actions satisfy all applicable statutory and regulatory requirements;
- the results of the agency's efforts to monitor specific actions the agency has authorized, funded or carried out, and the consequences of those decisions;
- and any feedback mechanism an agency has created to ensure that a program satisfies its statutory mandates, regulatory requirements, and applicable goals, and minimizes unintended consequences from the agency action.

If an agency proposes to satisfy its section 7(a)(2) obligations using a decision-making process that insures that listed resources are not exposed to specific actions without undergoing a tiered section 7 consultation on a specific action, we examine the evidence available to determine whether and to what degree the agency's decision-making process is likely to produce that outcome. If the agency's decision-making process is designed to mitigate the consequences of exposing listed resources to specific actions, we examine the evidence available to determine whether and to what degree the agency's decision-making process produces that outcome. When we consult on a pre-existing program, the program's general pattern of performance over its history becomes our primary evidence.

After we examine an agency's decision-making process, we then examine the classes of actions the program would authorize, fund, or carry out. This step of our assessment is designed to determine whether and to what degree listed resources are likely to be exposed to different classes of activities that would be authorized, funded, or carried out under a program. During this step of our assessment, we consider the geographic distribution, timing, and constraints of the different classes of activities that would be authorized, funded, or carried out by a program (the geographic distribution of the activities' effects defines the action area of programmatic consultations). These analyses represent the "exposure analyses" of our programmatic consultations in which we try to identify the populations or subpopulations, ages (or life stages), and gender of the individuals that are likely to be exposed to an action's effects.

Then we use the best scientific and commercial data available to identify the classes of intended and unintended consequences that are likely to result from the different classes of activities. These analyses identify the probable direct and indirect consequences of exposing listed resources to those classes of activities for listed individuals, populations, and species, and designated critical habitat; these analyses represent the "response analyses" and "risk analyses" of our programmatic consultations. Our "response analyses" review the scientific and commercial data available to determine whether, how, and to what degree listed resources are likely to respond given their exposure to the intended and unintended consequences of classes of activities. Our "risk analyses" begin by identifying the probable consequences of those responses for the "performance" of listed individuals, and then they identify the consequences of changes in individual performance on the viability of the populations those individual represent. Our "risk analyses" conclude by determining the consequences of changing the viability of the populations, and the species those populations comprise. As stated earlier, our assessment is based on the general patterns that we observe through our prior experiences with a program or class of activities.

Evidence Available for the Consultation

To conduct our analyses, we considered lines of evidence available through published and unpublished sources that represent evidence of adverse consequence or the absence of such consequences. In particular, we considered information contained in *EPA's Biological Evaluation for Cyanide*, and published information used in deriving the 304(a) aquatic life criteria for cyanide. We supplemented this information by conducting electronic searches of literature published in English or with English abstracts using research platforms in the *Online Computer Library Center's* (OCLC) *First Search*, *CSA Illumina*, *Toxline*, *Science Direct*, *Water Resources Abstracts*, *Oceanic Abstracts*, *BioOne Abstracts and Indexes*, *Conference Papers Index*, *Lexis-Nexis*, *Google Scholar*, and *ISI Web of Science*. These platforms allowed 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 *Basic Biosis*, *Dissertations*, *ArticleFirst*, *Proceedings*, *Aquatic Sciences and Fisheries Abstracts* and ECO databases, which index the major journals dealing with ecological risk, biology and ecology of particular species, and the toxicology of cyanide in freshwater, estuarine, and marine ecosystems (e.g., journals such as *Environmental Toxicology and Chemistry*, *Human and Ecological Risk Assessment*, *Journal of Mammalogy*, *Canadian Journal of Zoology*, *Transactions of the American Fisheries Society*, *Conservation Biology*, and others).

For our literature searches, we used paired combinations of the keywords cyanide, salmon, marine mammals, sea turtles, sturgeon, coral, sawfish, seagrass, and many others to search these electronic databases. Electronic searches have important limitations, however. First, often they only contain articles from a limited time span (e.g., First Search only provides access to master's theses and doctoral dissertations completed since 1980 and Aquatic Sciences and Fisheries Abstracts only provide access to articles published since 1964). Second, electronic databases commonly do not include articles published in small or obscure journals or magazines. Third electronic databases do not include unpublished reports from government agencies, consulting firms, and non-governmental organizations. To overcome these limitations, we supplemented our electronic searches by searching the literature cited sections and bibliographies of references we retrieved to identify additional papers that had not been captured in our electronic searches. We acquired references that, based on a reading of their titles and abstracts, appeared to comply with our keywords. If a references' title did not allow us to eliminate it as irrelevant to this inquiry, we acquired the reference.

Additionally, we separately searched the websites of the U.S. Geological Survey, EPA, states, U.S. Department of Health and Human Services, and the International Union for the Conservation of Nature (IUCN) for documents and data that identified potential effects of cyanide on marine, estuarine, and freshwater ecosystems and the individuals that inhabit these ecosystems. We conducted searches of EPA's Toxics Release Inventory (TRI) and Storage and Retrieval (STORET) databases for water quality data to identify areas where discharges are monitored for cyanide, and to characterize the general patterns of known occurrence and reported values over time and space.

From these documents we extracted the following: when the information for the study or report was collected, the study design, which species the study gathered information on, the sample size, the form of cyanide associated with the study, whether the study was conducted in a controlled laboratory environment or *in situ* (in the field or natural environment), whether other stressors were associated with study, study objectives, and study results. There is some concern that the exposure concentration and response observed in some studies on cyanide may not be accurate or reliable given differences between the analytical methods used, and forms of cyanide studied. Therefore, we followed a similar classification scheme as developed by Gensemer et al. (2007) to make comparisons among the type of cyanide exposure measurements performed in the studies. We classified studies according to whether they measured: (1) free cyanide using a reliable test method (e.g., ASTM 4282-95); (2) measured free cyanide but the test method accuracy is unknown; (3) measured weak acid dissociable cyanide; (4) measured total cyanide, and provided an estimate of free cyanide; (5) measured total cyanide, but did not estimate free cyanide; (6) did not provide an analytical verification of the cyanide concentration. Within each class of studies, we ranked each of the studies based on the quality of their study design, sample sizes, level of scrutiny before and during publication, and study results. We ranked carefully designed experiments (for example, experiments that control potentially confounding variables) higher than experiments that were not designed to control potentially confounding variables. We ranked carefully designed experiments higher than computer simulations, and we ranked studies on the response of listed species higher than studies on other, non-listed species. We also ranked studies that produced large sample sizes with small variances higher than studies with small sample sizes or large variances. Articles that did not rely on evidence produced by controlled experiment, uncontrolled field experiments, opportunistic observations of animal behavior or computer simulation received the lowest rating, but we considered the arguments and conclusions within these articles within our analyses.

Application of this Approach in this Consultation

The EPA proposes to continue approving state and tribal water quality standards for cyanide, which are based on their recommended 304(a) aquatic life criteria that were developed and published in the 1980s under the authority of the CWA. Section 304(a) of the CWA, the goals and purposes of the CWA, the implementing regulations for water quality standards (40 CFR 130-131), and the *Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses* (later referred to as "the *Guidelines*"; Stephan et al. 1985) form the foundation, or the standards that the cyanide criteria were designed to meet. This Opinion represents NMFS' evaluation of whether EPA's approval of state or tribal water quality standards, or federal water quality standards promulgated by EPA, that are identical to or more stringent than the section 304(a) aquatic life criteria for cyanide satisfies EPA's obligations pursuant to section 7(a)(2) of the ESA of 1973, as amended.

NMFS' evaluation proceeds by asking if the approval of cyanide consistent with (or more stringent than) the 304(a) aquatic life criteria for cyanide proposed by EPA is likely to prevent the exposure of endangered species, threatened species, and designated critical habitat from aqueous cyanide concentrations that are toxic, given the approach it uses to approve water quality standards? If listed resources are not likely to be exposed to the direct and indirect effects of cyanide from activities the water quality standards would authorize, both individually and cumulatively, given the approach EPA uses to approve a water quality standards, we would conclude that EPA's proposal to continue recommending the 304(a) aquatic life criteria for cyanide is not likely to jeopardize the continued existence of endangered species, threatened species, or result in the destruction or adverse modification of designated critical habitat under NMFS' jurisdiction. If, however, listed resource are likely to be exposed to the direct and indirect effects of cyanide from activities the water quality standards would authorize, both individually and cumulatively, we would ask whether and to what degree listed species are likely to respond to their exposure, given the approach EPA uses to approve a water quality standards. As part of this analysis, we would examine whether and to what degree EPA has identified chemical, physical and biological scenarios that influence cyanide toxicity and presence in the environment inhabited by listed species and their critical habitat, the nature of any in situ effects, and the consequences of those effects for listed resources under NMFS' jurisdiction, to determine if EPA can insure that the approval of state and tribal water quality standards that they are proposing is not likely to jeopardize the continued existence of endangered species or threatened species, or result in the destruction or adverse modification of critical habitat that has been designated for these species.

Understanding the Water Quality Program

EPA has asked that the Services consult on their approval of water quality standards where states and tribes adopt the standards that are consistent with or more stringent than the nationally recommended 304(a) aquatic life criteria. Since our analysis must consider the direct and indirect effects of the action together with the direct and indirect effects of any interdependent and interrelated actions[3], a critical first step to any consultation is determining whether and to what extent there are actions interrelated and interdependent with the action under consultation.

While EPA's BE does not examine interrelated and interdependent actions, it did provide us partial insight into the issue of what EPA considers interrelated and interdependent actions, inasmuch as EPA highlighted the general protective measures that states may adopt as part of their water quality programs as further evidence that listed resources would rarely, if ever, be exposed to cyanide at the recommended criteria values. Since the action as EPA has described it in its BE and subsequent documents, is the approval of water quality standards that states and tribes implement as enforceable standards for cyanide then it follows that the direct and indirect effects of any actions that are interrelated or interdependent with that approval must be considered in this consultation.

We developed a simple conceptual model to illustrate our understanding of the overall water quality program, and to assist us in determining whether there are actions that are interrelated or interdependent to the EPA's recommended 304(a) aquatic life criteria and subsequent approval of cyanide standards when states and tribes adopt the recommended numeric values. In part, we were interested in exploring the relationships among program components and EPA's decision to approve a particular standard and, specifically, whether the protective measures described in the BE and imposed by states and tribes should be considered in this consultation as interrelated and interdendent with the action to approve.

Our model depicts the relationship between EPA's 304(a) aquatic life criteria and other components of EPA's water quality-based approach to pollution control (Figure 1). Figure 1 also illustrates those relationships between "any action authorized, funded or carried out by" EPA under the composite program and section 7(a)(2) of the ESA. The model is based on the discussion of the water quality-based approach to pollution control, and the interrelated parts of executing the CWA as it was described by EPA in the *Water Quality Standards Handbook* (EPA 1994), information on the program characteristics that were provided by EPA in the cyanide BE, and is also based on our prior experiences with state water quality standards and NPDES permits issued by states and EPA. Our model, as with any descriptive model, represents a simplified map of the characteristics of the larger water-quality based pollution control program.

The goals and policies of the CWA establish the foundation for EPA's pollution control program. Pollution control begins, in part, with the identification of a target or priority pollutant and EPA's decision to "develop and publish" … (and from time to time thereafter revise) 304(a) criteria for water quality for that particular pollutant. EPA derives 304(a) aquatic life criteria through an established decision-making process outlined by the *Guidelines* (Stephan et al. 1985), which we depict at the top of Figure 1. Upon deriving a numeric value for a pollutant, EPA recommends (publishes) the numeric value for adoption and implementation. Publication typically involves a draft stage and a final stage in between which EPA solicits public comments.

The national aquatic life criteria provide the foundation for a wide variety of programs aimed at

[3] Interdependent actions are those actions that have no independent utility apart from the action under consideration. Interrelated actions are those actions that are part of a larger action and depend upon the larger action for their justification (50 CFR 402.02).

addressing pollution control under the CWA. EPA's 304(a) aquatic life criteria serve as guidelines or recommendations to states and tribes for defining water column concentrations of cyanide that EPA expects would protect against adverse ecological effects to aquatic life in fresh and salt water. The 304(a) aquatic life criteria recommendations are calculated to protect aquatic organisms from unacceptable toxicity during acute (short) and chronic (long) exposures in the water column. The intent is to define a level in the waterbody of a pollutant that will be fully protective of the designated uses of a water body and that a state or tribe may use in adopting its regulatory water quality standards and achieve the goals of their waterbodies (BE page 11, 40 CFR 131.2). States and tribes may use the 304(a) aquatic life criteria as a basis for developing enforceable water quality standards. The CWA requires all states to adopt water quality standards to restore and maintain the physical, chemical, and biological integrity of the Nation's waters. The CWA allows that states with an approved water quality program may adopt the 304(a) criteria as an enforceable standard (in combination with other relevant program elements), or they may modify the recommended criteria to reflect site-specific conditions, or create unique water quality standards (40 CFR 131.11(b)).

The focus of our national consultation with EPA, are those instances where a state or tribe "adopts" a water quality standard that is consistent with the recommended aquatic life criteria. In Figure 1, the consultation on this national approval is depicted by the yellow box, "National Section 7 Consultation".

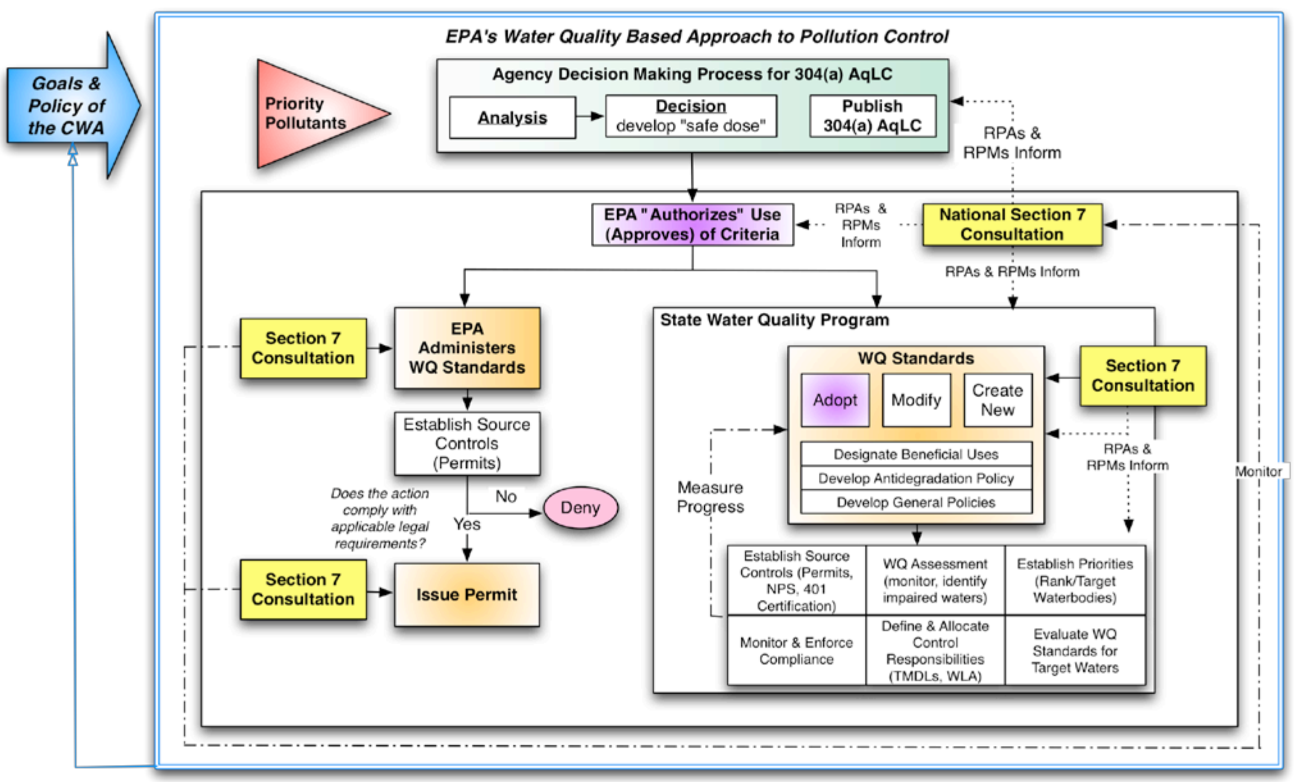

Figure 1. EPA's 304(a) aquatic life criteria and its relationship to the water quality-based pollution control program and section 7.

An approved standard, however, is more than just a numeric value for pollutants. Rather “a water quality standard defines the water quality goals of a water body, or portion thereof, by designating the use or uses to be made of the water and by setting the criteria necessary to protect the uses. States adopt water quality standards to protect public health or welfare, enhance the quality of water and serve the purposes of the Clean Water Act….. Such standards serve the dual purposes of establishing the water quality goals for a specific water body and serve as the regulatory basis for the establishment of water-quality based treatment controls and strategies….. (40 CFR 131.2).” A state’s water quality program contains eight general parts with specific regulatory requirements and guidance. We included the eight general parts of a state’s water quality program on the right side of Figure 1. The eight parts are described by EPA (1994) as follows:

Establish protection levels. EPA’s approach to pollution control begins with the identification of problem water bodies, and the water quality standards establish the assessment goals, and the water body uses intended for protection. Standards are not simply a numeric pollutant threshold level, but standards consist of three main elements (1) designated beneficial uses of a waterbody or segment of a waterbody (e.g., protection of aquatic life, recreation), (2) water quality criteria necessary to protect the use or uses of that particular waterbody (expressed in either numeric or narrative form[4]), and (3) an antidegradation policy. Additionally, states, at their discretion, may adopt general policies in their standards affecting the application and implementation of standards (e.g., mixing zone policies, variance policies, critical flow policies for permit based-limits).

Water quality assessments. Once water quality standards are adopted, states conduct water quality monitoring to identify those waters that are “water quality limited” or not meeting standards. Monitoring is important to evaluating whether designated uses are attained, determining whether Total Maximum Daily Limits (TMDL) are needed, and assessing compliance with permits and so on. Under section 305(b) of the CWA states are required to prepare a water quality inventory every two years to document the status of assessed water bodies. At this point the state may make a determination that the water body is not impaired but that the condition is due to natural conditions.

Establish priority waterbodies. When waters are identified that don’t meet standards or are water quality limited, a state is expected to prioritize (rank) waterbodies for TMDL development.

Evaluate water quality standards for target waters. At this point in the water quality management process, States have targeted priority water quality-limited water bodies. EPA recommends that States re-evaluate the appropriateness of the water quality standards for the targeted waters if: 1) States have not conducted in-depth analyses of appropriate uses and criteria; 2) changes in the uses of the water body may require changes in the standard; 3) more recent water quality monitoring show the standard is being met; and, 4) site-specific criteria may be appropriate because of specific local environmental conditions or the presence of species more or less sensitive than those included in the national criteria data set.

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[4] States must adopt numeric standards for toxic pollutants listed pursuant to section 307(a)(1) of the CWA and for which criteria have been published under 304(a).

1. *Define and allocate control responsibilities.* For water quality limited waters, States must establish a total maximum daily load (TMDL) that quantifies pollutant sources, and a margin of safety, and allocates allowable loads to the contributing point and non-point source discharges so that the water quality standards are attained. EPA recommends States develop TMDLs on a watershed basis.

2. *Establish source controls.* Source loads of pollutants are controlled through the TMDL, waste load allocations (WLA), best management practices (BMPs), and through the technology-based and water quality-based controls implemented through the NPDES permitting process. Although, many states and territories have authority to implement at least a portion of the NPDES program in their jurisdiction, EPA retains full or partial authority in many states and territories. In the case of nonpoint sources, both State and local laws may authorize the implementation of nonpoint source controls, such as best management practices (BMPs) or other management measures.

Monitor and enforce compliance. Monitoring is critical to the water quality-based decision making, and includes assessing compliance with TMDLs, permits, as well as in water loading (necessary to also capture nonpoint source pollution loads) and attainment of water quality goals. Point source dischargers are required to provide reports on compliance with NPDES permit limits. A monitoring requirement can be put into the permit as a special condition as long as the information is collected for the purposes of writing a permit limit. Effective monitoring programs are also required for evaluating nonpoint source control measures and EPA provides guidance in implementing and evaluating nonpoint source control measures. EPA and States are authorized to bring civil or criminal action against facilities that violate their NPDES permits. State nonpoint source programs are enforced under State law and to the extent provided by State law.

Measure progress. Arguably, one of the most important elements of the overall program are the efforts by the states (and EPA) to assess the effectiveness of the controls and standards, to determining water quality standards need to be revised, or more stringent controls are necessary (e.g., through permits or WLA and TMDLs). This is particularly important in determining whether a water body on the 303(d) list of impaired waters achieves water quality standards and can be removed from the state's 303(d) list, or to determine if WLA must be modified. This element is depicted as a feedback arrow between the general program elements and the foundation of state programs, the numeric standards and the policies that govern the program execution.

The left side of Figure 1 depicts those aspects of the water quality-based approach to pollution control that are approved and carried out directly by EPA. Criteria developed, published and approved by EPA are the foundation for many actions administered by EPA, including the promulgation of national water quality standards, and the issuance of NPDES permits.

Figure 1 also illustrates a general need by EPA to consult on actions that EPA "approves, funds, and carries out" under the program, which includes nationally approved criteria, as well as the approval of new state standards and the triennial review of those standards, and EPA's issuance of NPDES permits. The scope and details of such consultations depend upon EPA's discretionary control or authority to insure that its decisions on these actions comply with the CWA, its implementing regulations and policies. The yellow boxes in Figure 1 generally depict those areas where EPA would consult with the Services on their actions.

The consultation boxes in Figure 1 are linked to the national consultation to illustrate that NMFS will use the evidence obtained in regional and site-specific consultations to determine whether a particular consultation produced the expected results or produced results that were not consistent with the assumptions and conditions of the national consultation. That is, this first national consultation establishes a feedback framework to assist NMFS in assessing (a) the reliability, validity, or relevance of any evidence it relied upon in its national consultation; (b) whether the national consultation produced the anticipated results or produced results that were not consistent with subsequent consultations, (c) assessing the current status of any reasonable and prudent alternatives, reasonable and prudent measures, terms and conditions, and reporting requirements that EPA must comply with under the national consultation; and (e) the current and projected trends of listed resources, and the altered environmental baseline. The arrows in connecting these consultations in Figure 1 are broken because this is a newly developed feedback framework and has not previously been implemented by NMFS in its water quality consultations with EPA.

Interrelated and Interdependent Actions

The effects of EPA's 304(a) criteria recommendation must be understood in the larger context of the CWA. This larger context is framed by Congress' stated objective, goals, and policies of the CWA, and the programs and activities authorized by the CWA and implemented by EPA, and states and tribes to achieve these objectives, goals and policies. It is the CWA requirement that all states adopt water quality standards to restore and maintain the physical, chemical, and biological integrity of the Nation's waters that places the standards at the core of the overall strategy for water-quality based pollution control. As described previously, standards serve as the regulatory basis for the water quality-based approach to pollution control and are used to identify water quality problems caused by various land uses, such as improperly treated wastewater discharges, runoff or discharges from active or abandoned mining sites, sediment, and so on.

As a practical matter most states and tribes adopt EPA's recommended 304(a) criteria for most pollutants as part of their water quality standards even though they can develop unique criteria for their waters (EPA 1999). According to a review of state water quality criteria for cyanide, we found that more than 80% of the states and territories adopted EPA's acute and chronic freshwater criteria for cyanide or criteria that were more stringent[5] (Appendix A). Eleven states (Arizona, Arkansas, California, Iowa, Louisiana, Nebraska, Ohio, Oklahoma, Texas, Washington, and Wisconsin) adopted higher values in their standards, some significantly so. Some of these states adopted different values for cold waters versus warm waters (e.g., Arizona) or specified particular areas subject to these different values (e.g., Washington, California). States that set significantly higher standards than EPA's nationally recommended 304(a) criteria included Iowa, Louisiana, Ohio and Texas. No states set lower salt water values than EPA

[5] We interpreted "more stringent" to be a lower value that would lead to less cyanide in the water. Most states and territories that had set lower standards for cyanide were only a few tenths to hundredths lower than the value recommended by EPA.

recommended, but a few established higher values. California established levels as high as 10.0 µg/L CN for the saltwater instantaneous maximum and Texas set their chronic and acute saltwater criteria at 5.6 µg/L CN. Local exemptions in some state waters are much higher than these broader state limits. For instance, Illinois allows for 100 µg/L for acute exposure and 1,000,000 µg/L in some waterways in Cook County (home to Chicago). Although several states adopted new standards that differ from EPA's recommended values, the fact that most states follow EPA recommendations for cyanide verbatim illustrates the influence that EPA's guidance has on state standards. We suspect that EPA's action to develop and publish (recommend) 304(a) aquatic life criteria likely is sufficient to dissuade many states from investing the resources to develop unique water quality standards, particularly in times of economic hardship and reduced state budgets.

That the CWA creates an independent statutory requirement that states adopt enforceable water quality standards is sufficient reasoning to support the argument that state standards have "independent utility" and would not generally be considered interdependent with EPA's 304(a) criteria. However, since the vast majority of states adopt the 304(a) criteria as developed and published by EPA, and EPA has requested that this consultation, programmatically, address their need to consult on their approval of the water quality standards that are consistent with, or more stringent than the 304(a) recommended criteria the argument of independent utility is moot. That is, it is EPA's expectation that this national consultation address their general action to approve any state or tribal water quality standards for cyanide that are consistent with, or more stringent than the numeric value they recommend, and by doing so EPA hopes to eliminate subsequent regional consultations on water quality standards. .

As we described earlier, the level of protection afforded to a water body under the CWA is defined by the sum of the designated uses, criteria, antidegradation policy[6], and general policies. While all are required in a state submission, the designated uses and criteria are particularly inseparable components of a water quality standard as evidenced by EPA's language on approving a submission. That is, to approve a proposed water quality standard EPA must find that a state has adopted uses that are consistent with the requirements of the CWA and that adopted criteria protect those designated uses. EPA cannot approve a numeric value for a particular pollutant, like cyanide, if that numeric value does not support the uses the state has designated for a particular water body. The designated uses are integral to the approval and have no independent utility apart from the approval of a water quality standard, but are one of the most important parts of a water quality standard. More so, a water quality standard, by definition, is not complete without a finding that a particular criterion meets the designated uses. Therefore, designated uses are also interrelated with a particular criterion value because they are integral parts of the standard (part of the larger action), and depend upon the larger action for their justification.

[6] In a January 27, 2005, memorandum to it Regional Offices, EPA concluded that ESA section 7 consultation does not apply to EPA's approvals of state antidegradation policies because EPA's approval action does not meet the "Applicability" standard defined in the regulations implementing section 7 of the ESA (EPA 2005; 50 CFR 402.03). Section 402.03 of the consultation regulations (50 CFR part 402) states that section 7 and the requirements of 50 CFR part 402 apply to all actions in which there is discretionary Federal involvement or control. EPA concluded that they are compelled to approve State antidegradation policies if State submissions meet all applicable requirements of the *Water Quality Standards Regulation* (40 CFR part 131) and lack discretion to implement measures that would benefit listed species. As a result, EPA determined that consultation is not warranted on antidegradation policies because the Agency does not possess the regulatory authority to require more than the minimum required elements of the regulations.

When we embarked on this evaluation, however, we noted we were particularly interested in determining whether the protective measures described in the BE and imposed by states and tribes should be considered in this evaluation. EPA stated that states and tribes may, in addition to adopting numeric criteria, adopt: narrative criteria, biological criteria, or site-specific criteria for cyanide. EPA also noted that during implementation of their water quality standards, several other assumptions are made when allocating pollutants, for permitting purposes, among point source discharges to protection of species. As part of the TMDL and NPDES permit development, according to EPA most states and tribes use the following protective assumptions in the development of their TMDLs and water quality based effluent limitations: (1) assume that all dischargers are discharging the contaminant at the maximum permitted levels, (2) provide for an unallocated "margin of safety" when developing TMDLs, (3) assume the maximum permitted discharge volume, (4) assume the maximum concentration of loading of pollutants, (5) assume no environmental degradation of pollutants, (6) assume all discharged pollutants remain biologically available, (7) assume receiving stream flows are very low, (8) assume that acute toxicity limits apply at the "end of the pipe", (9) assume that only a portion of the design flow is available for mixing for controls on chronic toxicity, (10) assume that aquatic species live continuously at the "edge of the mixing zone", (11) assume no internal dilution of process wastewater, (12) assume conservative values for upstream concentrations of pollutants, (13) permit conditions should not be relaxed in subsequent permit reissuance (antibacksliding), (14) antidegradation requirements protect existing uses, (15) assume low threshold for "reasonable potential" if few data are available. While we cannot disagree that these components of a state's water quality program warrant further examination, and may even qualify as interrelated and interdependent actions that demonstrate the success (or failures) of various specific programs and the success of the overall water quality program, the BE provided no evidence of the general patterns of the implementation of these measures, nor an evaluation of the success or failures of these protective mechanisms across the national landscape. We further acknowledge that each TMDL, WLA, and NPDES could in fact be considered actions interdependent to EPA's approval of a state's water quality standards because the standards and goals for a water body "serve as the regulatory basis for the establishment of water-quality based treatment controls and strategies (40 CFR 131.2)."

Perhaps the most compelling reason that the above mentioned actions and other general program operations have independent utility, however, is the fact Congress intentionally divided many of these state and tribal actions into different sections of the CWA. In fact much of the statute directs the actions of state and tribes, not EPA's, supporting state autonomy for the protection of their waters. That the sections were designed to work together to achieve the goals set forth by Congress should not be a surprise, and in of itself should not be reason to consider all programs that rely on the water quality standards as interrelated or interdependent to the approval of water quality standards. Thus we default to the statutory construct, and the distinctive sections of the Act that instruct states and tribes on the execution and operation of their overall water quality program, as providing the strongest argument for independent utility.

Moveover, we note that the inclusion into this consultation of the myriad of such actions as dictated by the different programs that rely on water quality standards would easily make this national consultation untenable in short order. Thus, unless we can establish evidence of the general pattern in which the protective measures EPA noted in their BE are implemented across states and tribes (information which was not contained within the BE) then these assertions served little relevance to our analysis on the national scale. We further submit that individual NPDES permits, TMDLs, WLA, and other management aspects of a state's water quality program, while emanating from EPA's approved water quality standards, merit evaluation in subsequent consultations, where appropriate. Where EPA does not retain discretion, and such actions may affect listed resources, then states and tribes ought to seek a permit from the Services pursuant to section 10(a)(1)(b) of the ESA. We therefore propose that while each of the actions that are part of the overall approach to protecting aquatic life in waters of the United States are targeted to assessing compliance with standards and instituting change to achieve compliance through modification to allowable discharges or to the standards themselves, they have independent and significant roles in achieving the goals of the CWA. Consequently, they merit separate reviews as appropriate under the ESA. Such separate reviews can be linked through our conceptual model feedback links (see Figure 1), to assist NMFS and EPA in conducting holistic review of the effectiveness of the programs for protecting listed resources.

What we cannot separate on the basis of independent utility, however, as they are intimately a part of the action as EPA has proposed it, are the elements of a state or tribal water standard that must be included in each submittal to EPA for review and in order for EPA to approve said standard (see EPA 1994). As established in the foregoing discussion, these include: designated uses, criteria, antidegradation policy, and general policies. Hence, we address these other components as they are an essential part of any standard EPA approves, as interrelated and interdependent actions to EPA's approval of a numeric pollutant value in a state or tribal standard. These interrelated and interdependent actions are discussed in the *Effects of the Action* section of this Opinion.

Water Quality Standards

Water quality standards, as mentioned previously, are the mechanism by which protection levels for a water body are established. The water quality standards establish the assessment goals (e.g., numeric or narrative criteria), and the water body uses intended for protection. Whenever a state revises or adopts a new water quality standard such revised or new standard must be submitted to EPA for review. The water quality standard must include designated uses consistent with the provisions of section 101(a)(20 and 303(c)(2) of the CWA, the methods used and analyses conducted to support water quality standards revisions, water quality criteria sufficient to protect the designated uses, an antidegradation policy, certification that the water quality standards were duly adopted pursuant to state law, and general information that will aid the EPA in determining the adequacy of the scientific basis of the standards (40 CFR 131.6).

According to the CWA, the standards shall protect the public health or welfare, enhance the quality of water and serve the purposes of the Act, and shall be established taking into consideration their use and value for public water supplies, propagation of fish and wildlife, recreational purposes, and agricultural, industrial, and other purposes, and also taking into consideration their use and value for navigation. The phrase to "serve the purposes of the Act" as defined in 303(c) of the CWA, means that the water quality standards should meet the objectives of the Act "to restore and maintain the chemical, physical, and biological integrity of the Nation's waters." In order to achieve this objective Congress declared that---

(1) It is the national goal that the discharge of pollutants into the navigable waters be eliminated by 1985;

(2) It is the national goal that where ever attainable, an interim goal of water quality which provides for the protection and propagation of fish, shellfish, and wildlife and provides for recreation in and on the water be achieved by July 1, 1983;

(3) It is the national policy that the discharge of toxic pollutants in toxics amounts be prohibited…."

These three goals, which are commonly referred to as the "zero discharge" goal, "the fishable/swimmable" goal, and the "no toxics in toxic amounts" goal, are accompanied in the statute by a number of subsidiary goals and policies (Adler et al. 1993). Water quality standards for aquatic life are primarily designed to meet the fishable/swimmable goal of the CWA.

Water quality standards (in particular, the numeric criteria coupled with a water body's designated uses) are the core mechanism for meeting the goal of the CWA, and "getting water quality standards right starts with getting designated uses right (EPA 2008a)." When a state submits a water quality standard, EPA must review and approve (or disapprove) a standard based upon whether a state has: (a) adopted uses that are consistent with the requirements of the CWA, (b) adopted criteria that protect the designated uses, (c) followed legal procedures for adopting standards, (d) whether the submission meets the regulatory requirements (40 CFR 131.5). In specifying appropriate water uses, each state must take into consideration the protection and propagation of fish, shellfish and wildlife, and recreation in and on the water (the "fishable/swimmable" goal among other things; 40 CFR 131.10(a)), whether or not a use is currently being attained.

Designated Uses

Designated uses are statements of management objectives and expectations for water bodies under state or tribal jurisdiction. As defined in 40 CFR 131.3, designated uses are specified in the water quality standards for each water body or water body segment regardless of whether or not they are being attained. Designated uses include, but are not limited to: water supply (domestic, industrial and agricultural); stock watering; fish and shellfish uses (salmonid migration, rearing, spawning, and harvesting; other fish migration, rearing, spawning, and harvesting); wildlife habitat; ceremonial and religious water use; recreation (primary contact recreation; sport fishing; boating and aesthetic enjoyment); and commerce and navigation.

The water quality standards regulation requires that states and tribes specify which water uses are to be achieved and protected. These uses are determined by considering the value and suitability of water bodies based on their physical, chemical, and biological characteristics as well as their geographical settings, aesthetic qualities and economic attributes. Each water body does not necessarily require a unique set of uses. Rather, water bodies sharing characteristics necessary to support a use can be grouped together. If water quality standards specify designated uses of a lower standard than those that are actually being attained, the State or Tribe is required to revise its standards to reflect these uses.

Antidegradation

Antidegradation implementation procedures identify the steps and questions that must be addressed when proposed activities may affect water quality. The water quality standards regulation requires that states and tribes establish a three-tiered antidegradation program. The specific steps to be followed depend upon which tier or tiers apply. These tiers are listed below:

- Tier 1: These requirements are applicable to all surface waters. They protect existing uses and water quality conditions necessary to support such uses. These uses can be established if they can be demonstrated to have actually occurred since November 28, 1975, or if water quality can be demonstrated to be suitable for such uses. If an existing use is established, it must be protected even if it is not listed in the water quality standards as a designated use.
- Tier 2: These requirements maintain and protect "high quality" water bodies where existing conditions are better than those necessary to support CWA § 101(a)(2) "fishable/swimmable" uses. Although the water quality in these water bodies can be lowered, states and tribes must identify procedures that must be followed and questions that must be answered before a reduction in water quality can be allowed. The water quality of these water bodies cannot be lowered to a level that would interfere with existing or designated uses.
- Tier 3: These requirements maintain and protect water quality in outstanding national resource waters (ONRWs) and generally include the highest quality waters of the United States. ONRW classification also offers special protection for waters of exceptional ecological significance. Except for certain temporary changes, water quality cannot be lowered in these waters. states and tribes decide which water bodies qualify as ONRWs.

In a January 27, 2005, memorandum to it Regional Offices, EPA concluded that ESA section 7 consultation does not apply to EPA's approvals of state antidegradation policies because EPA's approval action does not meet the "Applicability" standard defined in the regulations implementing section 7 of the ESA (EPA 2005; 50 CFR 402.03). Section 402.03 of the consultation regulations (50 CFR Part 402) states that section 7 and the requirements of 50 CFR part 402 apply to all actions in which there is discretionary Federal involvement or control.

EPA concluded that they are compelled to approve State antidegradation policies if State submissions meet all applicable requirements of the *Water Quality Standards Regulation* (40 CFR part 131) and lack discretion to implement measures that would benefit listed species. As a result, EPA determined that consultation is not warranted on antidegradation policies because the Agency does not possess the regulatory authority to require more than the minimum required elements of the regulations. For these reasons, antidegradation will not be a part of this consultation.

General Policies

States and tribes may adopt general policies and provisions regarding the implementation of water quality standards. These policies and provisions are subject to EPA review and approval. General policies must relate to designated use criteria or antidegradation. These policies and provisions include:

1. Mixing Zones: A mixing zone is a defined area surrounding or downstream from a point source discharge where the effluent is diluted by the receiving water and criteria otherwise applicable to the water body may be exceeded. At their discretion, states and tribes may allow mixing zones for point source discharges. Mixing zone procedures describe the methodology for determining the location, size, shape, and quality of mixing zones.

2. Variances: Variances temporarily relax a water quality standard. They are subject to public review every three years and may be extended. A variance may specify interim water quality criteria applicable for the duration of the variance. States or tribes may wish to include a variance as part of a water quality standard as an alternative to removing a designated use. Variances are intended to help assure that further progress toward improving water quality is achieved.

3. Low Flows: State and tribal water quality standards may identify policies and procedures to determine critical low flow conditions. For example, such procedures are applied when calculating discharge requirements to be included in National Pollutant Discharge Elimination System (NPDES) permits.

Evaluating Exposure at the National Level

The next step in our analysis involved evaluating the contaminant, cyanide (the stressor), in the environment in which the listed resources occur. Although we searched, we simply could not find sufficient data to conduct a quantitative assessment of the likelihood of exposure, or the likelihood of exposure at a particular numeric value. Therefore, our analysis focuses largely on the consequences of an exposure at criterion value. However, to examine a species' (and their critical habitat's) risk of exposure, we searched for evidence that would help us describe the (1) the transport, fate, and persistence of cyanide in the environment, (2) the distribution of uses and occurrences of cyanide across the U.S., and (3) temporal and spatial changes, where we could find evidence of these changes, across the U.S.

We began by constructing a simple conceptual model for evaluating the effects of contaminants on listed species and critical habitat. This model depicts the release of a contaminant into the environment, its transport through the environment and its contact with the listed species (Figure 2). The fate of pollutant, and whether it reaches habitats containing listed species, depends upon a wide number of variables including chemical form and structure, volume dispersed and the manner in which it is dispersed, distance of travel, and processes of sorption, degradation, and dilution, to name a few.

In describing the basic properties of cyanide, we also looked at chemical, biological and physical attributes in the environment that might act as "filters" or "magnifiers" that influence the relationship between cyanide and the induction of effects on listed species. For instance, Cloern (2001) used tidal energy to illustrate the importance of filters in eliciting certain responses within an ecosystem—tidal energy influences turbulent kinetic energy and mixing in shallow waters, and ultimately the expression of eutrophication. Differences in tidal amplitude are one mechanism by which different estuaries will respond dissimilarly to equally high loads of nutrients, and in turn the filters acting within different ecosystems would dictate potentially very different pollutant concentrations to which listed species would be exposed.

Some of the particular features of an ecosystem or site that can act as filters, influencing the nature, magnitude, and spatial and temporal distribution of pollutants to which listed species may be exposed include: water hardness, pH, precipitation, wind, light, bathymetry, stratigraphy, topography, trace gas absorption, mineral weathering, elemental storage ability, soil chemical processes, microbial transformation, and so on. For site-specific assessments, as much as possible, the site's features should be described and used to evaluate associations between the listed species and their critical habitat, and the particular pollutant under evaluation. At the national scale, however, we look for evidence of the types of filters that generally would be expected to interact with cyanide along its general transport pathway, and that would influence its availability, toxicity and severity.

Our simple transport model, illustrated in Figure 2, serves as a map for our analysis. That is, it illustrates the main pathways— the physical course cyanide generally takes from the source to the receptor organism or communities of interest (Suter et al. 2002). For section 7 evaluations of pollutants, the receptor organism is the listed species or designated critical habitat. An exposure pathway is complete when the chemical(s) under evaluation reach the receptor organism. A pathway is incomplete when the stressor does not reach the organism under evaluation. Simply, in the latter case when the pathway is incomplete, the chemical does not co-occur with the listed species or its designated critical habitat.

Our conceptual transport model emphasizes the exposure route through surface waters because the primary route of exposure to chemical contaminants for most of NOAA's trust resources will often be through water-borne exposures. As with any conceptual model, this visual depiction of exposure pathways is a simplified representation of what can be expected in the natural environment. For instance, not only would some species be exposed to surface water contaminants, animals that live portions of their life cycle out of water like many marine mammals (aquatic-dependent species) may be exposed to contaminants on land. Even wholly aquatic species, like salmon may be exposed to contaminants in terrestrial vegetation—through leaf litter and insects (allochthonous stream input)—and contaminated soils that enter the aquatic environment.

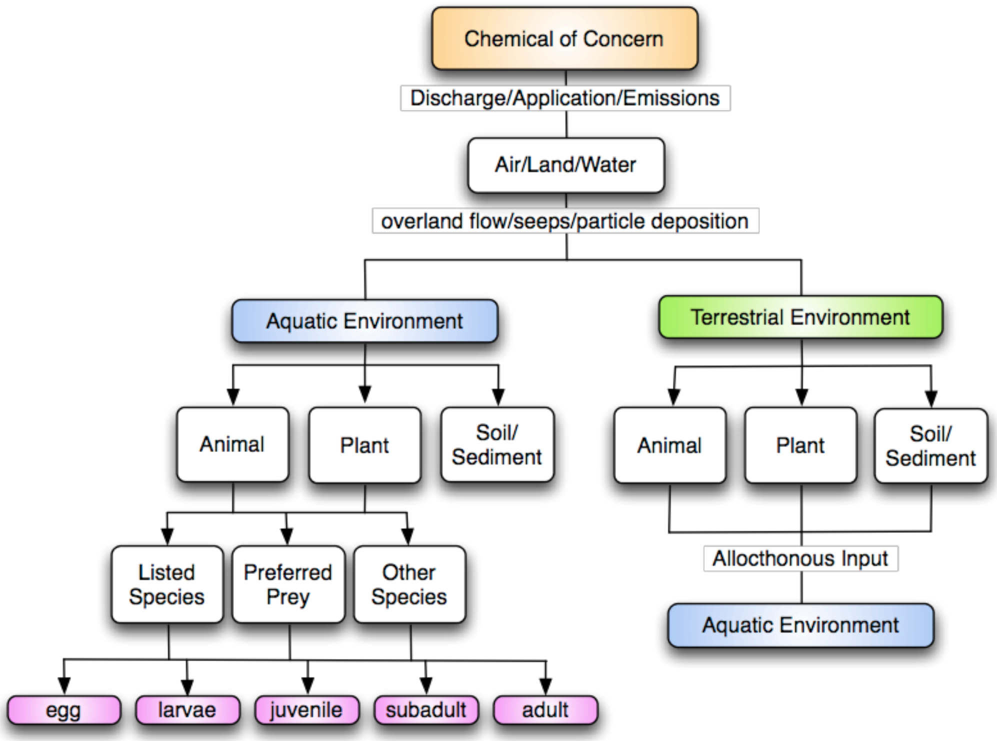

Figure 2. Simple transport model depicting pollutant pathways to aquatic habitats and aquatic species.

We would consider an exposure pathway complete when the chemical under evaluation would generally be expected to reach the listed species and incomplete when the stressor does not reach the listed species. Often the more difficult aspect of a section 7 evaluation is identifying the indirect pathways by which a listed species or their critical habitat is affected by a chemical stressor, which requires an examination of relationship of the listed species to the communities of which it is a part, and the environment in which it resides, depends upon, and is adapted. To capture indirect exposure pathways we look at the relationship of the listed species to the community and environment in which it lives. This means, that not only do we look for effects of cyanide directly acting on the listed species, we examine the effect that cyanide has on the biological community and environment in which the listed species lives (Figure 3). We do this to determine if cyanide would induce community and environmental changes that would likely affect the listed species, such as changes in prey availability or health.

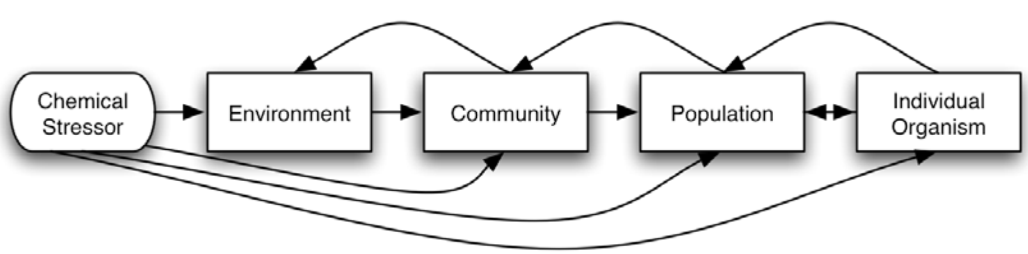

Figure 3. A chemical stressor and its potential relationships with organisms in the wild

Our challenge in this step is to identify: what populations, life history forms or stages are exposed to the proposed action; the number of individuals that are exposed; the pathways of their exposure; the timing and duration of their exposure; the frequency and magnitude of the concentrations of the exposure; and how exposure might vary depending upon the characteristic of the environment and individual behavior. Typically, in this step of our analysis we would identify how many individuals are likely to be exposed, which populations the individuals represent, where and when the exposure would occur, how long the exposure would occur, the frequency of the exposure, and any other particular details that help characterize the exposure. To do this we require knowledge of a species' population structure and distribution, migratory behaviors, life history strategy, and abundance.

All of the species under NMFS' jurisdiction are "aquatic" or "aquatic-dependent", meaning that at least one or more life stages are aquatic and could potentially be exposed to aqueous pollutants. Therefore, since EPA has asked that this consultation cover their national approval of standards that are consistent with their recommended aquatic life criteria, we began our assessment with the basic assumption that all of the listed species and critical habitat under NMFS' jurisdiction, as well as any species proposed for listing and critical habitat proposed for listing, would potentially be exposed to cyanide at the recommended criteria values at some time during their life cycle. NMFS assumes the recommended criteria value is an appropriate starting assumption for exposure in particular because the recommended value is assumed to represent a "safe dose" of cyanide.

Using this assumption, we asked whether and to what degree would animals that are exposed at the recommended level be protected if exposed at that value (this is part of our response analysis). Next, we asked whether and to what degree the proposed action and any interrelated and interdependent actions would mitigate, minimize or avoid allowing cyanide discharges to reach (or exceed) the recommended criteria. Because this examination is done at the national level, we looked for general patterns of cyanide where that information was available to us. We used such information as general patterns of the distribution of uses, manufacturing, and incidental occurrences of cyanide in the environment, and we looked for temporal and spatial changes in these uses to characterize the past 20 years of cyanide in the environment, and as a basis for predicting the future of cyanide in the environment across our action area. Our evaluation is explained in detail in our effects analysis.

Action Area

EPA has defined the action area for the cyanide consultation, and for the 304(a) aquatic life criteria consultations in general as all "waters of the United States" including "territorial seas" (see the BE, pages 8 and 9, and the *Methods Manual* page 6). The CWA (33 USC 1362) defines territorial seas as "the belt of the seas measured from the line of ordinary low water along that portion of the coast which is in direct contact with the open sea and the line marking the seaward limit of inland waters, and extending seaward a distance of three miles." This action area includes such waters within and surrounding Indian Country, the 50 States, and all United States territories. The terms "waters of the United States" is defined under 40 CFR Section 122.2 and reiterated in EPA's cyanide BE.

As early as 1789, the United States territorial sea was established at three nautical miles. On 27 December 1988, however, President Regan issued a proclamation that extended the United States territorial sea to 12 nautical miles from the baselines of the United States. Although, nothing in the proclamation extended or otherwise altered existing federal or State law subsequent to the 1988 proclamation, several federal laws adopted the terms of the Proclamation to define the United States territorial sea for purposes of that particular statute (e.g., the Nonindigenous Aquatic Nuisance Prevention and Control Act of 1990, the Antiterrorism and Effective Death Penalty Act of 1996). However, others, including the Federal Water Pollution Control Act (aka. the CWA) continue to use the three mile limit in its definition of the United States territorial sea.

The action area for the purposes of consultation, however, is not limited to the area of an agency's jurisdiction. Rather, in consultation the action area is defined as all areas to be affected directly or indirectly by the federal action and not merely the immediate area involved in the action (50 CFR 402.02). Many federal actions that NMFS consults on occur in the United States territorial sea, the contiguous zone, exclusive economic zone, and on the high seas. The issue of jurisdiction is relegated to the point in the Opinion at which NMFS prescribes management actions (Reasonable and Prudent Alternatives and Reasonable and Prudent Measures) for the purpose of exempting the taking of threatened and endangered species from the prohibitions of section 4(d) and 9 of the ESA. (See the section of this Opinion titled *Reasonable and Prudent Alternatives*). Consequently, the action area for EPA's 304(a) aquatic life criteria consultations includes the minimal area, as defined by the freshwater, estuarine and ocean water bodies of the United States and its territories (delineated by the CWA) and any areas the particular pollutant under consultation (in this case cyanide) is transported beyond these limits by such biotic and abiotic factors as river runoff, tidal energy, topography, stratigraphy, biota [trapping/assimilation), that may influence chemical transport processes beyond original areas of dispersion. We expect, based on the chemical processes (sources, transport, and fate) of cyanide, which are described later in this Opinion, that most of the action area for this consultation on cyanide is contained by the jurisdictional waters as outlined by the CWA. However, in certain localities we expect that conveyance systems may extend to the outer edge of this action area, or that the discharge plume may extend beyond three nautical miles. Unfortunately, we cannot identify the specific areas or conveyance systems where this may occur, and thus recognize that our action area is generally delineated according to three nautical miles extending from the United States coastline.

Since NMFS has jurisdiction over marine and anadromous threatened and endangered species, and their critical habitat, this Opinion addresses the potential effects of EPA's aquatic life criteria in a portion of the action area defined for 304(a) aquatic life criteria. Specifically, this Opinion focuses on the direct and indirect effects of the recommended criteria along the coastal regions of the United States and its territories, where listed resources under NMFS' jurisdiction occur. As such, although interior fresh waters (e.g., landlocked lakes and ponds of the midwest United States) constitute a portion of the action area for this consultation, listed resources under NMFS' jurisdiction do not occur in these areas and these portions of the action area are not considered further in this Opinion.

Status of the Species and Critical Habitat

In this section of this Opinion we describe the threatened and endangered species and their designated critical habitat that occur in the action area and may be exposed to EPA's approved aquatic life criteria for cyanide. All listed species within NMFS' jurisdiction are "aquatic" or "aquatic dependent" and may occur within portions of the action area for the recommended aquatic life criteria. NMFS has determined that the following species and critical habitat may occur within the action area for EPA's 304(a) aquatic life criteria for cyanide (Table 2).

Table 2. Species Listed as Threatened and Endangered and Proposed for listing, and their designated Critical Habitat (denoted by asterisk) in the Action Area. Double asterisks denote Proposed Critical Habitat.

Common Name (Distinct Population Segment or Evolutionarily Significant Unit)	Scientific Name	Status
Cetaceans		
Beluga whale** (Cook Inlet)	*Delphinapterus leucas*	Endangered
Blue whale	*Balaenoptera musculus*	Endangered
Bowhead whale	*Balaena mysticetus*	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
Pinnipeds		
Hawaiian monk seal*	*Monachus schauinslandi*	Endangered
Steller sea lion (Eastern*)	*Eumetopias jubatus*	Threatened
Steller sea lion (Western*)		Endangered
Marine Turtles		
Green sea turtle (Florida & Mexico's Pacific coast colonies)*	*Chelonia mydas*	Endangered
Green sea turtle (All other areas)*		Threatened
Hawksbill sea turtle*	*Eretmochelys imbricate*	Endangered
Kemp's ridley sea turtle	*Lepidochelys kempii*	Endangered
Leatherback sea turtle* (also **)	*Dermochelys coriacea*	Endangered
Loggerhead sea turtle	*Caretta caretta*	Threatened
Olive ridley sea turtle (Mexico's Pacific coast breeding colonies)	*Lepidochelys olivacea*	Endangered
Olive ridley sea turtle (All other areas)		Threatened
Anadromous Fish		
Atlantic salmon*	*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*)		Threatened
Green sturgeon (Southern*)	*Acipenser medirostris*	Threatened
Gulf sturgeon*	*Acipenser oxyrinchus desotoi*	Threatened
Shortnose sturgeon	*Acipenser brevirostrum*	Endangered
Smalltooth sawfish*	*Pristis pectinata*	Endangered
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*)		Endangered
Steelhead (Upper Columbia River*)		Threatened
Steelhead (Upper Willamette River*)		Threatened
Marine Invertebrates		
Black abalone	*Haliotis cracherodii*	Endangered
Elkhorn coral*	*Acropora palmata*	Threatened
Staghorn coral*	*Acropora cervicornis*	Threatened
White abalone	*Haliotis sorenseni*	Endangered
Marine Plants		
Johnson's seagrass*	*Halophilia johnsonii*	Threatened
Proposed for Listing		
Bocaccio	*Sebastes paucispinis*	Proposed Endangered
Canary rockfish	*Sebastes pinniger*	Proposed Threatened
Pacific eulachon/smelt	*Thaleichthys Pacificus*	Proposed Threatened
Spotted seal	*Phoca largha*	Proposed Threatened
Yelloweye rockfish	*Sebastes ruberrimus*	Proposed Threatened

The species' narratives that follow focus on attributes of a species' life history and distribution that influence the manner and likelihood that a particular species may be exposed to the proposed action, as well as the species potential response and risk when exposure occurs. Consequent narratives summarize a larger body of information on worldwide distribution, as well as localized movements within fresh water, estuarine, intertidal, and ocean waters, population structure, feeding, diving, and social behaviors. We also provide a brief summary of the species status and trends as a point of reference for our jeopardy determinations, which we make later in this Opinion. That is, we rely on a species' status and trend to determine whether or not an action's direct or indirect effects are likely to increase the species' probability of becoming extinct. Similarly, each species narrative is followed by a description of its critical habitat with particular emphasis on any essential features of the habitat that may be exposed to the proposed action and may warrant special attention. Because this is a national consultation that does not consider site-specific data, we only summarize information on the geographic distribution of the species, their ecological relationship with waters of the United States, their status, and the principal threats to their survival and recovery.

Species Not Considered Further in This Opinion

Species and Critical Habitat under Joint Jurisdiction

The Services share joint jurisdiction for the management of sea turtles, gulf sturgeon, Atlantic salmon. For sea turtles, NMFS is responsible for their in-water conservation while FWS is responsible for their conservation on land. This Opinion discusses the effects of the proposed action on listed marine sea turtles and their designated critical habitat in the following section.

The Services have divided the consultation responsibilities for Atlantic salmon according to whether the federal action occurs in fresh water or estuarine or marine waters (74 FR 29344). When a federal action traverses marine and fresh waters, then the Services decide which agency will assume the lead role for consultation. For the purposes of this consultation, the FWS' Opinion addresses the effects of the action on Atlantic salmon pursuant to section 7. However, because Atlantic salmon are one of the few species for which direct exposure data are available on the effects of cyanide, this Opinion contains numerous references to this data and its utility in evaluating the effects of cyanide on other species. The full evaluation as to how the federal action affects Atlantic salmon, and whether the action is likely to jeopardize the continued existence of Atlantic salmon is addressed in the FWS' Biological Opinion on cyanide. Similarly, NMFS and FWS share jurisdiction over Gulf sturgeon and generally divide consultations according to whether the federal activity occurs within marine or fresh water. The critical habitat listing for gulf sturgeon clarifies, however, that the FWS will consult with EPA on water quality issues (68 FR 13370). Therefore, the FWS' Biological Opinion on cyanide addresses whether the federal action is likely to jeopardize the continued existence of gulf sturgeon, and the likelihood that the designated critical habitat would be destroyed or adversely modified.

Species and Critical Habitat Not Likely to be Adversely Affected by the Proposed Action

Based upon our analysis, we established that we can concur with EPA's effect determination that a number species are not likely to be adversely affected when exposed to cyanide at criterion values. Specifically, we would not expect the following threatened or endangered species to respond physically, physiologically, or behaviorally to cyanide at the CMC or the CCC: Blue whale, bowhead whale, fin whale, humpback whale, North Atlantic right whale, North Pacific right whale, sei whale, sperm whale, Hawaiian monk seal, Western Steller sea lion, Eastern Steller sea lion, green sea turtle, hawksbill sea turtle, Kemp's ridley sea turtle, leatherback sea turtle, olive ridley sea turtle, smalltooth sawfish, elkhorn coral, staghorn coral, white abalone, black abalone, and Johnson's seagrass. Similarly, we expect the designated critical habitat for the following species is not likely to be adversely affected by cyanide at the CMC or the CCC: North Pacific right whale, Hawaiian monk seal, Western Steller sea lion, Eastern Steller sea lion, green sea turtle, hawksbill sea turtle, leatherback sea turtle, smalltooth sawfish, elkhorn coral, staghorn coral, and Johnson's seagrass. Based upon our analysis, the following proposed species[7] are not likely to be adversely affected when exposed to cyanide at the salt water CMC or the CCC: bocaccio, canary rockfish, spotted seal and yelloweye rockfish. The effects of the proposed action on the Pacific eulachon have not been evaluated.

Listed cetaceans, pinnipeds, sea turtles, marine invertebrates and plants, and marine fishes are distributed in coastal areas that may be exposed to aquatic cyanide. Certain species, like the blue whale and sei whale, are likely to have limited exposure to cyanide sources as their migratory patterns are circumglobal with definite seasonal movements to offshore areas outside the likely extent of cyanide discharges. Nonetheless, we could not conclude that exposures would not occasionally occur, and thus evaluated the potential responses of these species when exposed to cyanide levels equivalent to the salt water CCC and CMC.

Unfortunately, data to evaluate the potential responses of listed marine species or for suitable surrogate species when exposed to cyanide at the recommended aquatic life values is severely lacking. It is for these reasons that Gensemer et al. (2007) declined to evaluate the protectiveness of the saltwater cyanide criteria for marine threatened and endangered species. Pursuant to Section 7 of the ESA, however, we are not proffered the opportunity to withhold judgment. To evaluate the effects of cyanide, particularly on marine species, the lack of data is disconcerting and warrants studies to evaluate response thresholds for more marine species.

In the interim, until further investigations that establish threshold responses are available, current information suggests that the effects of cyanide at the salt water CMC and CCC values of 1.015 µg CN/L on listed marine species and their designated critical habitat, and proposed marine species are extremely unlikely to occur and thus discountable. Our conclusion is based on available data on the responses of marine species relative to the saltwater aquatic life criteria thresholds. The recommended saltwater CMC and CCC are set at very low levels, 1.015 µg CN/L. The CMC value for cyanide was driven by data on the eastern rock crab, *Cancer irroratus*. The species mean acute value for eastern rock crab is 4.893 µg CN/L making the crab six times more sensitive than the next most sensitive marine species, the calanoid copepod, *Acartia tonsa* (EPA 1985). Data were available on the chronic effects of cyanide to only two marine species when EPA established the recommended aquatic life criteria, the mysid, *Mysidopsis bahia*, and the sheepshead minnow, *Cyprinodon variegatus*. Recognizing that these species are relatively resistant to cyanide, EPA set the CCC equal to the CMC because doing so was probably more indicative of the chronic sensitivity of the rock crab than obtained using chronic response data from other species and using other derivation methods (ACR). We found no data to suggest that listed marine species would respond to cyanide exposures at or below 1.015 µg CN/L.

Marine Mammals & Turtles

According to the *Methods Manual*, marine mammals and sea turtles are part of a broad category of "aquatic-dependent" species that whose respiratory oxygen is gained from surface air, not from oxygen dissolved in the water column (like "aquatic species"). For these species, the

7 Proposed species were listed after the completion of EPA's BE. Little data exists to discern adverse effects at levels below the saltwater CCC or CMC. Unlike the other proposed species, the Pacific Eulachon has a freshwater and saltwater life stage. Salt water exposure to cyanide at the CCC and CMC is not likely to result in adverse effects; however, Pacific eulachon still to be evaluated consistent with the approach used to evaluate the effects of the action on other freshwater fishes.

analysis would focus primarily on dietary exposure because this route is generally considered the important route of exposure. The *Methods Manual* expressly discounts dermal or other routes of exposure as areas that are "not explicitly sought in the literature search" when EPA develops the biological evaluations for pollutants but notes that in the event information is uncovered during a literature search that would suggest otherwise, it would be considered in EPA's effects analysis. Otherwise, the assessment of toxicity on aquatic-dependent listed species, which accounts for all listed marine mammals, sea turtles, and pinnipeds, is based on the estimated dietary effects concentration (dietary EC). The dietary effect would be evaluated by producing estimates of bioconcentration factors (BCFs) and bioaccumulation factors (BAFs). However, there is no published evidence to suggest that cyanide bioaccumulates in fresh- or saltwater aquatic animals. As such exposure to cyanide via the dietary or sediment pathways may not be particularly important.

High doses of cyanide that are ingested can be rapidly lethal (doses exceeding the saltwater CCC), and low doses of cyanide are rapidly metabolized and excreted. Eisler (1991) suggested that repeated sublethal dietary doses may be tolerated by many species for extended periods. The acute oral toxicity of cyanide was calculated on a small set of surrogate species and based on the wet weight of the oral dose. Species used for this analysis ranged from a variety of birds to small and large mammals such as rats, and cows. The minimum acute dietary LD_{50} for birds is 1.4 mg/kg body mass and for mammals is 2.2 mg/kg body mass. Based on these values, marine mammals, sea turtles, and pinnipeds would have to consume cyanide well in excess of the saltwater CMC to experience a lethal response. The saltwater CMC is also likely set below any potential chronic dietary threshold for marine mammals and turtles.

EPA also evaluated toxicity values for a wide range of food items, grouping them into common categories (e.g., insects, invertebrates, fish, etc). Calculated response values were above the CMC and the CCC for both saltwater and freshwater environments. Although the central tendency of the response value was used for the assessment, and not the 5^{th} percentile conservative estimate as was used for listed species, we expect this approach provides a reasonable estimate of adverse effects to prey species particularly given that most of NMFS' species are generalist feeders and a minor reduction in a particular food item should generally result in discountable and insignificant effects to listed species. For instance, the fin whale is a baleen whale and eats krill, a tiny crustacean. As mentioned previously, the species most sensitive to cyanide is the eastern rock crab. The threshold values from the eastern rock crab were used to determine the effect that cyanide may have on krill. Similarly, the loggerhead sea turtle feeds on mollusks, sponges and crabs. The food item analysis conducted by EPA for this species, was driven by the EC for mollusks (4.7) but should have been reviewed against the invertebrate EC (2.2), because it eats invertebrates and mollusks the dietary analysis should have been reviewed against the lowest EC possible. Nonetheless, the outcome remains the same in this instance—that is, marine food items should not be adversely affected by cyanide at the saltwater criteria.

Based on the best scientific and commercial data available, as discussed previously, we do not expect that the proposed action would adversely affect the quantity, quality or availability any of the constituent elements of critical habitat, or the physical, chemical, or biotic phenomena that give the designated area value for the conservation of the species when no constituent elements were identified in the designation. Although through the proposed action, we would expect critical habitat for North Pacific right whale, Hawaiian monk seal, Western Steller sea lion, Eastern Steller sea lion, green sea turtle, hawksbill sea turtle, the leatherback sea turtle, and proposed critical habitat for the leatherback sea turtle would be exposed to cyanide, the concentration of cyanide would be sufficiently low that we expect the effects would be discountable. As reviewed in the above summary, there is little evidence to discern the effects of cyanide at levels as low as recommended by EPA in the saltwater aquatic life criteria. That said, the data that is available suggests that 1.015 µg CN/L is not likely to adversely alter water quality that supports growth and development, feeding and food resources, reproduction, areas for nesting and reproduction, or other physical, chemical or biological attributes of critical habitat for these species.

Marine Invertebrates and Plants

No dose-response data is available to derive a lethal threshold for *Acropora* species. Much of the data on corals is largely from studies that have examined the effects of the very destructive practice of cyanide fishing, which tends to employ cyanide concentrations well in excess of the saltwater criteria. At high doses, cyanide kills coral, causes loss of zooxanthellae, impaired photosynthesis, disruption of protein synthesis and altered rates of mitosis (Jones and Steven 1997; Jones and Hoegh-Guldberg 1999; Cato and Brown 2003; Cervino 2003). A few studies have been conducted on the short-term exposure of coral species to sublethal concentrations, but the concentrations have been well above the saltwater criteria. According to Dzombak et al. (2006) some studies have observed no response of coral to cyanide exposures at concentrations as low as 26 µg CN/L. More research is needed to discern the response threshold for listed species. However, given the limited data available at this time, it appears that exposure to cyanide at the low concentrations recommended by the aquatic life criteria that that any effects would likely be discountable and insignificant.

We also have very little data to suggest what the threshold response concentrations would be for marine plants. Evidence suggests that some plants are capable of transforming cyanide through enzymatic activity and can avoid cyanide intoxication by directly degrading the cyanogenic compounds or assimilating them into their metabolism. The effectiveness of this response would depend upon the plant, the balance of activity and the exposure concentration. EPA's best estimate of response thresholds is based on the freshwater blue-green algae, *Microcystis aeruginos*, and the marine red algae. The latter has a NOEC of 11 µg CN/L, well above the saltwater CMC or CCC. Using red algae as a surrogate to predict the response of Johnson's sea grass, we expect the effects of cyanide at the aquatic life criteria would be discountable and insignificant.

There were too few data available to generate a species sensitivity distribution for white or black abalone through the class level. We found only one study on the effect of cyanide on an abalone species, the *Haliotis varia*, the varied ear shell or variable abalone. Given that the varied ear shell abalone is within the same genus, the reported LC_{50} of 1012 µg CN/L is the best estimate of a lethal response for both black abalone and white abalone. Lasut (1999) studied the effects of cyanide and salinity on the mortality of abalone and found that mortality increased within decreased salinity. Abalone subjected to lethal concentrations of potassium cyanide and sodium cyanide experienced a 19% increase risk of mortality when exposed to 25‰ salinity over that observed in 34‰ salinities. Even so, the response occurs well above the saltwater CMC. Therefore, we would not expect the species would be adversely affected when exposed to cyanide at the CMC saltwater value of 1.015 µg CN/L.

Based on the best scientific and commercial data available, as discussed previously, we do not expect that the proposed action would adversely affect the quantity, quality or availability any of the constituent elements of critical habitat, or the physical, chemical, or biotic phenomena that give the designated area value for the conservation of the species when no constituent elements were identified in the designation. Although through the proposed action, we would expect critical habitat for elkhorn coral, staghorn coral, and Johnson's seagrass would be exposed to cyanide, the concentration of cyanide would be sufficiently low that we expect the effects would be discountable. As reviewed in the above summary, there is little evidence to discern the effects of cyanide at levels as low as recommended by EPA in the saltwater aquatic life criteria. That said, the data that is available suggests that 1.015 µg CN/L is not likely to adversely alter water quality that supports growth and development, feeding and food resources, reproduction, areas for nesting and reproduction, or other physical, chemical or biological attributes of critical habitat for these species.

Marine Fishes

Too few data exist to generate a species sensitivity distribution estimate for this smalltooth sawfish, or the recently proposed rockfish species, bocaccio, yelloweye, and canary rockfish, through the class level. In comparison of the mean LC50 and NOEC values for the most closely related marine fishes range from 59.3 to 372 and 5.608 to 35.18 µg CN/L, respectively. Data on most acutely sensitive marine fish, the Atlantic silverside, results in acute and chronic EC_As of 26.12 and 5.608 µg CN/L, in the range of the most acutely sensitive freshwater fish species. Since insufficient data are available to model species sensitivity distributions for marine species, we relied on the calculated EC_As of the most sensitive marine fish for which data was available in making our effects determination. Although not included in EPA's biological evaluation, the three proposed rockfish would be evaluated using the same EC_A values, as not enough data exists to employ other evaluation methods. As such, data on the Atlantic silverside suggests that the saltwater cyanide criteria would likely result in discountable and insignificant effects on bocaccio, yelloweye, and canary rockfish, and smalltooth sawfish.

Based on the best scientific and commercial data available, as discussed previously, we do not expect that the proposed action would adversely affect the quantity, quality or availability any of the essential features of critical habitat. Although through the proposed action, we would expect critical habitat for smalltooth sawfish would be exposed to cyanide, the concentration of cyanide would be sufficiently low that we expect the effects would be discountable. As reviewed in the above summary, there is little evidence to discern the effects of cyanide at levels as low as recommended by EPA in the saltwater aquatic life criteria. That said, the data that is available suggests that 1.015 µg CN/L is not likely to adversely alter water quality that supports growth and development, feeding and food resources, reproduction, areas for nesting and reproduction, or other physical, chemical or biological attributes of critical habitat for these species.

Species and Critical Habitat Likely to be Adversely Affected by the Proposed Action

Anadromous Fishes

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 1991). In this section, we discuss the distribution, status, and critical habitats of the nine species[8] of endangered and threatened Chinook salmon separately, and summarize their common dependence on waters of the United States. However, because Chinook salmon in the wild are virtually indistinguishable between listed species, and are the same biological species we begin this section describing those characteristics common across ESUs (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 56ºN (Healy 1991). Within each race, there is often variation in age at seaward migration, age of maturity, timing of spawning migrations, male precocity, and female fecundity.

The general Chinook salmon life cycle spans fresh and marine waters, with one reproductive event per adult (that is, 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). However, 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 (Healy 1991).

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 they 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

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

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.

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

Once emerged, fry behavior varies among populations and among individuals within races. Some juvenile Chinook salmon rear in fresh water for a few weeks to a few years, others move immediately downstream 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 1991). Generally, Chinook salmon outmigrants (termed smolts) are about 2 to 5 inches long when they enter saline (often brackish) waters. The process of smoltification is a physiologically demanding process that enables salmon to adapt to sea water and maintain the appropriate osmotic pressure necessary to maintain body fluid concentration and composition, and homeostasis as the fish enters waters of increased salinity. The transformation from the fresh water fry/parr juvenile stage to smolt involves multiple physiological changes including an increase 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, the ratio of weight standardized to length (condition factor) declines and total body lipid content declines (Wedemeyer et al. 1980). Several factors can affect smoltification process, not only at the interface between fresh water and salt water, but higher in the watershed as the process of transformation begins long before fish enter salt waters including: exposure to chemicals such as 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 fresh water after 2 to 4 years at sea. Fishery catches indicate that ocean- and stream-type fish exhibit divergent migratory pathways while in the ocean (Healey 1983, 1991). 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, 1991).

Chinook salmon feed on a variety of prey organisms depending upon life stage. Adult oceanic Chinook salmon eat small fish, amphipods, and crab megalops (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 chironomid larvae and pupae, *Daphnia*, *Eogammarus*, *Corphium* and *Neomysis*, as well as juvenile herring, sticklebacks and other small fish. In fresh water, Chinook salmon juveniles feed on adult and larval insects including terrestrial and aquatic insects such as dipterans, beetles, stoneflies, chironomids, and plecopterans (Healey 1991).

Threats

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 species.

Anthropogenic Threats. Salmon survive only in aquatic ecosystems and, therefore, depend on the quantity and quality of those ecosystems. 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).

Salmon along the west coast of the United States share many of the same threats. Therefore, anthropogenic threats for all species and populations are summarized here. 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 broodstock for artificial propagation, the genetic erosion of populations (introgression, hybridization), an increased incidence of disease in the wild 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 (Reinsenbichler 1997; Mundy 1997). Targeted fishing of larger individuals results in excluding the most fecund individuals from spawning (Reinsenbichler 1997). Genetic changes that promote smaller body sizes have occurred in heavily exploited populations in response to size-selective harvest pressures (Reinsenbichler 1997; Mundy 1997; Swain et al. 2007). Fishing pressure can reduce age at maturity in fished populations as the fished populations compensate for the reductions in the numbers of spawning adults (Reinsenbichler 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, particularly PCBs, DDT and its congeners, pesticides, and endocrine disruptors are of particular concern. 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 salt water systems (Varanasi et al. 1993). 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. 1993).

Climate change poses significant hazards to the survival and recovery of salmonids along the west coast. Paleoecological data (which exclude anthropogenic influences) suggest regional and global climate factors on decadal, centennial, and millennial time scales are tied to abundance patterns of Pacific salmonids (Finney et al. 2009). Increases in global temperatures are likely to have profound effects on salmonids directly and indirectly through altered hydrological regimes. Increases in instream temperatures may decrease habitat available for refugia, increase species interactions and competition, accelerate incubation timing and premature emergence, increase susceptibility to parasites and disease, reduce fry survival, delay migration and spawning, and accelerate loss of energy reserves. Using emission scenarios from the Intergovernmental Panel on Climate Change (IPCC), O'Neal (2002) estimates that direct thermal changes in freshwater temperatures could cause the loss of between 4-20% of existing salmon and trout habitat by the year 2030, 7-34% by 2060, and 14-42% by 2090, depending on the trout or salmon species, IPCC emission scenario considered, and the model used. Projected salmon habitat loss would be most severe in Oregon and Idaho, at losses of 40% or greater of 2007 habitat estimates. While the predicted losses are substantial, the estimates may underestimate the overall effect global climate change will have on salmon and trout abundance since these models do not consider the related effects from changes in seasonal hydrological patterns and water volumes that result from altered weather patterns and precipitation (O'Neal 2002).

Changes in hydrological regimes are closely linked to salmon abundance (Hicks et al. 1991; Clark et al. 2001). From studies that have examined the effects of timber harvest and other changes in land use patterns, we know that changes in hydrology (i.e., increased peak flows, decreased low flows, altered timing discharge events, and rapid fluctuations in flows) can profoundly affect salmon abundance and the amount and availability of quality habitat. Hydrology is strongly correlated to in-redd and young of the year survival, can lead to the displacement of young fish, alter immigration and emigration timing, alter the volume of available habitat by affecting channel structure (e.g., pool to riffle ratios, debris loading, substrate composition, erosion and sediment loading) and the relative abundance of salmon and trout species within a watershed, as well as the relative abundance of age-classes (see Hicks et al. 1991; Gregory and Bisson 1997). Such ecosystem changes are also likely to alter macroinvertebrate communities and habitats, affecting important forage for salmon and trout (McCarthy et al. 2009; Williams et al. 2009).

Upstream changes in riverine habitat can affect downstream estuarine ecosystems through alterations in sediment delivery (timing and volume), and changes in freshwater volumes and timing can influence the volume of the spring/summer salt-wedge (O'Neal 2002). In turn, changes in the trophic dynamics of the estuary may occur. At the same time, 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. 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. 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 return of adult migrations into coastal water and rivers.

California Coastal Chinook Salmon

Distribution and Description of the Listed Species

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, California. 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 3 identifies populations within the California Coastal Chinook salmon ESU, their abundances, and the relative contribution of artificially propagated fish to the population.

Table 3. 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 *in* Good et al. 2005). The species exists as small populations with highly variable cohort sizes. The Russian River probably contains some natural production, but the origin of those fish is not clear because of a number of introductions of hatchery fish over the last century. The Eel River contains a substantial fraction of the remaining Chinook salmon spawning habitat 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. 2005).

Long-term trends in Freshwater Creek are positive, and in Canyon Creek, although only slightly different than zero, the trend is positive (Table 3). Long-term trends in Sprowl and Tomki creeks (tributaries of the Eel River), however, are negative. Good et al. (2005) 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.

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.

Central Valley Spring-Run Chinook Salmon

Distribution and Description of the Listed Species

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.

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.

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; Clark 1929).

Table 4. 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.

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. 2005). Before construction of Friant Dam, nearly 50,000 adults were counted in the San Joaquin River alone (Fry 1961). 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. 2005).

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. 2005). 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. All three spring-run Chinook salmon populations in the

Central Valley have long-and short-term positive population growth. 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.

Factors contributing to the downward trends in this ESU include: reduced access to spawning/rearing habitat behind impassable dams, climatic variation, water management activities, hybridization with fall-run Chinook salmon, predation, and harvest (CDFG 1998). Several actions have been taken to improve and increase the primary constituent elements of critical habitat for spring-run Chinook salmon, including improved management of Central Valley water (e.g., through use of CALFED Environmental Water Account and Central Valley Project Improvement Act (b)(2) water accounts), implementing new and improved screen and ladder designs at major water diversions along the mainstem Sacramento River and tributaries, removal of several small dams on important spring-run Chinook salmon spawning streams, and changes in ocean and inland fishing regulations to minimize harvest. Although protective measures and critical habitat restoration likely have contributed to recent increases in spring-run Chinook salmon abundance, the ESU is still below levels observed from the 1960s through 1990.

Threats from hatchery production (i.e., competition for food between naturally spawned and hatchery fish, and run hybridization and homogenization), climatic variation, reduced stream flow, high water temperatures, predation, and large scale water diversions persist.

Lower Columbia River Chinook Salmon

Distribution and Description of the Listed Species

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. Seventeen artificial propagation programs are part of this ESU: The Sea Resources Tule, Big Creek Tule, Astoria High School (STEP) Tule, Warrenton High School (STEP) Tule, Elochoman River Tule, Cowlitz Tule, North Fork Toutle Tule, Kalama Tule, Washougal River Tule, Spring Creek National Fish Hatchery Tule, Cowlitz spring (Upper Cowlitz River and Cispus River), Friends of the Cowlitz spring, Kalama River spring, Lewis River spring, Fish First spring, and the Sandy River Hatchery Chinook salmon 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.

Lower Columbia River Chinook salmon have three life history types, including 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 salmon typically enter the Columbia River in August through October to spawn in the mainstem of the large rivers (Kostow 1995). Spring Chinook salmon enter freshwater in March through June to spawn in upstream tributaries and generally emigrate from fresh water 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). 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 Chinook salmon population is the healthiest and has a reasonable probability of being self-sustaining. Abundances largely declined during 1998 to 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 5).

Most populations for which data are available have a long-term declining population trend (Table 5). 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 to 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-run Chinook salmon 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 Chinook salmon populations have not seen increases in recent years as pronounced as those that have occurred in many other geographic areas.

Table 5. 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)

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 Rivers 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 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. 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 Lower Columbia River Chinook salmon. 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.

Upper Columbia River Spring-run Chinook Salmon

Distribution and Description of the Listed Species

The Upper Columbia River spring-run Chinook salmon ESU includes all naturally spawned populations of Chinook salmon 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. Six artificial propagation programs are part of this ESU: the Twisp River, Chewuch River, Methow Composite, Winthrop National Fish Hatchery, Chiwawa River, and White River spring-run Chinook salmon 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. Spring-run Chinook salmon currently spawn in only three river basins above Rock Island Dam: the Wenatchee, Entiat, and Methow Rivers. Table 6 identifies the Upper Columbia River Chinook salmon ESU populations, their abundances, and estimates of the proportion of hatchery fish that contribute to the run size.

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. After migration, Upper Columbia River spring-run Chinook salmon hold in freshwater tributaries until spawning in late summer, peaking in mid- to late August. Juvenile spring-run Chinook salmon remain in fresh water for a full year before emigrating to salt water in the spring of their second year.

Table 6. 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 to 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% to 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 as 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 decline 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).

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 to 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 bankfull 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 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 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.

Puget Sound Chinook Salmon

Distribution and Description of the Listed Species

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, including rivers and streams flowing into Hood Canal, South Sound, North Sound and the Strait of Georgia in Washington. Twenty-six artificial propagation programs are part of the ESU: the Kendal Creek Hatchery, Marblemount Hatchery (fall, spring yearlings, spring sub-yearlings, and summer run), Harvey Creek Hatchery, Whitehorse Springs Pond, Wallace River Hatchery (yearlings and sub-yearlings), Tulalip Bay, Issaquah Hatchery, Soos Creek Hatchery, Icy Creek Hatchery, Keta Creek Hatchery, White River Hatchery, White Acclimation Pond, Hupp Springs hatchery, Voights Creek Hatchery, Diru Creek, Clear Creek, Kalama Creek, George Adams Hatchery, Rick's Pond Hatchery, Hamma Hamma Hatchery, Dungeness/Hurd Creek Hatchery, and Elwha Channel Hatchery Chinook salmon 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.

The Puget Sound ESU is comprised of 31 historical populations, of which 22 or more are believed to be extant and nine are considered extinct. Table 7 identifies the current populations within the Puget Sound Chinook salmon ESU for which there are data, and their recent abundance and 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 Chinook salmon spawners described by Healey (1991). However, within these generalized behavioral forms, significant variation occurs in residence time in fresh water and estuarine environments. For example, Hayman et al. (1996) described three juvenile Chinook salmon life histories with varying residency times in the Skagit River system in northern Puget Sound. Chinook salmon utilize nearshore Puget Sound habitats year-round, although they can be far from their natal river systems (Brennan et al. 2004).

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

Population	Historical Abundance[a]	Mean Number of Spawners (Natural-origin)[b]	Percent Hatchery Contribution (Range)[c]	λ (+/- SE)[d]
Nooksack-North Fork	26,000	1,538 (125)	91 (88-95)	0.75 (0.07)
Nooksack-South Fork	13,000	338 (197)	40 (24-55)	0.94 (0.05)
Lower Skagit	22,000	2,527 (2,519)	0.2 (0-0.7)	1.05 (0.09)
Upper Skagit	35,000	9,489 (9,281)	2 (2-3)	1.05 (0.06)
Upper Cascade	1,700	274 (274)	0.3	1.06 (0.05)
Lower Sauk	7,800	601 (601)	0	1.01 (0.12)
Upper Sauk	4,200	324 (324)	0	0.96 (0.06)
Suiattle	830	365 (365)	0	0.99 (0.06)
Stillaguamish-North Fork	24,000	1,154 (671)	40 (13-52)	0.92 (0.04)
Stillaguamish-South Fork	20,000	270		0.99 (0.02)*
Skykomish	51,000	4,262 (2,392)	40 (11-66)	0.87 (0.03)
Snoqualmie	33,000	2,067(1,700)	16 (5-72)	1.00 (0.04)
North Lake Washington		331		1.07 (0.07)*
Cedar		327		0.99 (0.07)*
Green		8,884 (1,099)	83 (35-100)	0.67 (0.06)*
White		844		1.16 (0.06)*
Puyallup	33,000	1,653		0.95 (0.06)*
Nisqually	18,000	1,195		1.04 (0.07)*
Skokomish		1,392		1.04 (0.04)*
Dosewallips	4,700	48		1.17 (0.10)*

Duckabush		43		
Hamma Hamma		196		
Mid Hood Canal		311		
Dungeness	8,100	222		1.09 (0.11)*
Elwha		688		0.95 (0.11)*

[a]Estimated total historical abundance for this ESU was about 700,000 fish, but is not meant to reflect a summation of individual river historic estimates. Individual river estimates of historical abundance are based on an EDT analysis as reported in Good et al. 2005.
[b]5-year geometric mean number of spawners (hatchery plus natural) for years 1998-2002. Geometric mean of natural origin spawners noted in parentheses. From Good et al. 2005.
[c]Percent hatchery-origin from 1997-2001. Estimates are from the TRT database and reported in Good et al. 2005.
[d]Short-term median population growth rate estimates assume that the reproductive success of naturally spawning hatchery fish is equivalent to that of natural origin fish. Except estimates noted * where an estimate of the fraction of hatchery fish is not available then λ represents hatchery fish + natural-origin spawners. Data years used for calculation 1990-2002 (Good et al. 2005).

Status and Trends

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. 2005). The disproportionate loss of early-run life history diversity represents a significant loss of the evolutionary legacy of the historical 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) (data years 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. 2005). 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. 1996). 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.

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 Puget Sound Chinook salmon 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. 2005).

Critical Habitat

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 bankfull elevation.

The designation for this ESU includes sites necessary to support one or more Chinook salmon 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 (5th field Hydrological Units) 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.

Sacramento River Winter-Run Chinook Salmon

Distribution and Description of the Listed Species

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. 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 8). 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 8. 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.

2005). Most recent estimates indicate that the short term trend is 0.26, while the population growth rate is still less than 1 (Table 8). 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. 2005).

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. 2005).

Snake River Fall-Run Chinook Salmon

Distribution and Description of the Listed Species

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. Four artificial propagation programs are part of this ESU: The Lyons Ferry Hatchery, Fall Chinook salmon Acclimation Ponds Program, Nez Perce Tribal Hatchery, and Oxbow Hatchery fall-run 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.

Historically, the primary fall-run Chinook salmon spawning areas occurred on the upper mainstem Snake River (Connor et al. 2005). 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 the gravels 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. 2005). 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. 2005). Migration downstream appears to be influenced by flow velocity within both river and reservoir systems (Tiffan et al. 2009).

Status and Trends

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 to 1949 was 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 to 1967) and the lower Snake River dams (1961 to 1975). Counts of natural-origin adult Snake River fall-run Chinook salmon 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. 2005). Numbers of natural-origin Snake River fall-run Chinook salmon 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.

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 to 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. Harvest impacts on Snake River fall Chinook salmon 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 fall Chinook salmon release groups. Hatcheries stocking fish to the Snake River fall run 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.

Critical Habitat

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 Snake River salmon (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 to Snake River fall-run Chinook salmon. Limiting factors identified for Snake River fall-run Chinook salmon 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 2005a).

Snake River Spring/Summer-Run Chinook Salmon

Distribution and Description of the Listed Species

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. Fifteen artificial propagation programs are part of the ESU: The Tucannon River conventional Hatchery, Tucannon River Captive Broodstock Program, Lostine River, Catherine Creek, Lookingglass Hatchery Reintroduction Program (Catherine Creek), Upper Grande Ronde, Imnaha River, Big Sheep Creek, McCall Hatchery, Johnson Creek Artificial Propagation Enhancement, Lemhi River Captive Rearing Experiment, Pahsimeroi Hatchery, East Fork Captive Rearing Experiment, West Fork Yankee Fork Captive Rearing Experiment, and the Sawtooth Hatchery spring/summer-run Chinook salmon 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. The Interior Columbia Basin Technical Recovery Team has 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 9 identifies extant populations within the Snake River spring/summer Chinook salmon ESU, their abundances, and the relative contribution of hatchery fish.

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 the following 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 9. Snake River spring/summer Chinook salmon populations and selected measures of population viability

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
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

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 to 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 to 1996. The 2002 to 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 60% of the 2002 return originated in hatcheries (Good et al. 2005). Good et al. (2005) 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. Although the average abundance in the most recent decade is more abundant than the previous decade, there is no obvious long-term trend.

Critical Habitat

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 to Snake River spring/summer Chinook salmon. 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 2006c).

Upper Willamette River Chinook Salmon

Distribution and Description of the Listed Species

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. Seven artificial propagation programs are part of the ESU: The McKenzie River Hatchery, Marion Forks/North Fork Santiam River, South Santiam Hatchery in the South Fork Santiam River, South Santiam Hatchery in the Calapooia River, South Santiam Hatchery in the Mollala River, Willamette Hatchery, and Clackamas hatchery spring-run Chinook salmon 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.

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 the 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. Upper Willamette River Chinook salmon enter the Columbia River and 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

NMFS originally listed Upper Willamette River Chinook salmon as threatened on March 24, 1999 (64 FR 14308), and reaffirmed their status as threatened on June 28, 2005 (70 FR 37160). The total abundance of adult spring-run Chinook salmon (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). Until recent years, interpretation of abundance levels has been confounded by a high but uncertain fraction of hatchery-produced fish. Although the number of adult spring-run Chinook salmon crossing Willamette Falls is in the same range (about 20,000 to 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, several of which exceed 70 percent (Good et al. 2005). The Calapooia River is estimated to contain 100 percent 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 this ESU: the spring Chinook salmon in the McKenzie River. Unfortunately, recently short-term declines in abundance suggest that this population may not be self-sustaining (Myers et al. 1998; Good et al. 2005). 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 survival rates in the ocean 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 to 40% of total historical habitat are no longer accessible because of dams (Good et al. 2005). 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

NMFS designated critical habitat for Upper Willamette River Chinook salmon on September 2, 2005 (70 FR 52630). Critical habitat for upper Willamette River Chinook salmon 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 bankfull 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 critical habitat for the Upper Willamette River Chinook salmon ESU, 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 to Upper Willamette River Chinook salmon. 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.

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 the Soviet Union 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 our 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 so they die after one spawning event. Their general life cycle spans fresh and marine waters, although chum salmon are more marine oriented than the other Pacific salmon, in that they spend very little time rearing in fresh water. Chum salmon spend 2 to 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.

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).

The time necessary for egg incubation until emergence of alevins in fresh water 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). This ocean-type migratory behavior contrasts with the stream-type behavior of other species in the genus Oncorhynchus (e.g., coastal cutthroat trout, steelhead, coho salmon, and most types of Chinook and sockeye salmon, exception pink salmon), which usually migrate to sea at a larger size, after months or years of freshwater rearing. Because of their small size chum salmon will form loosely aggregated schools, presumably to reduce predation by swamping predators (Miller and Brannon 1982; Pitcher 1986).

Generally, chum fry emigrate to estuaries between March through May where they forage on epibenthic and neritic food resources. The timing of juvenile entry into sea water is commonly correlated with nearshore warming and associated plankton blooms (Groot et al. 1991). 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). 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).

Threats

Natural Threats. Chum salmon are exposed to high rates of natural predation each stage of their life stage, and in particular during migration. Mortality at emergence or prior to emergence is significant because eggs develop in the interstitial spaces in the stream gravel, and storm surges that redeposit gravels 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.

Anthropogenic Threats. Chum salmon, like the other listed 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 fresh water, estuarine, and coastal ecosystems throughout the Pacific Northwest. These threats for are summarized in detail under Chinook salmon.

Columbia River Chum Salmon

Distribution and Description of the Listed Species

The Columbia River chum ESU includes all naturally spawned populations of chum salmon in the Columbia River and its tributaries in Washington and Oregon. Three artificial propagation programs are part of the ESU: The Chinook River (Sea Resources Hatchery), Grays River, and Washougal River/Duncan Creek chum 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.

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).

Table 10. 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).

Status and Trends

NMFS listed Columbia River chum salmon as threatened on March 25, 1999, and reaffirmed their threatened 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 16 historical populations, 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. 2005). 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.

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. Finally, most Columbia River chum populations have been functionally extirpated or are presently at very low abundance levels.

Historically, the 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. 2005).

Critical Habitat

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 bankfull 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 Columbia River chum salmon ESU, three subbasins were rated as having a medium conservation value, no subbasins were rated as low, and the majority of subbasins (18), were rated as having a high conservation value to Columbia River chum salmon. The major factors limiting recovery for Columbia River chum salmon 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.

Hood Canal Summer-Run Chum Salmon

Distribution and Description of the Listed Species

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) 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 Quilcene National Fish Hatchery, Hamma Hamma Fish Hatchery, Lilliwaup Creek Fish Hatchery, Union River/Tahuya, Big Beef Creek Fish Hatchery, Salmon Creek Fish Hatchery, Chimacum Creek Fish Hatchery, and the Jimmycomelately Creek Fish Hatchery summer-run chum hatchery programs. 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 11 identifies populations within the Hood Canal summer-run chum salmon ESU, their abundances, and hatchery input.

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 summer-run chum salmon generally migrate through Hood Canal and into the main body of Puget Sound.

Status and Trends

NMFS listed Hood Canal summer-run chum salmon as threatened on March 25, 1999 (64 FR 14508), and reaffirmed as threatened on June 28, 2005 (70 FR 37160). Historically, Hood Canal summer-run chum salmon comprised an estimated 16 populations. Only eight extant populations remain within this ESU (Good et al. 2005). 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 many population remain adult returns for some populations showed modest improvements in 2000, with upward trends continuing in 2001 and 2002. The recent 5-year mean abundance is variable among populations in the species, 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 the 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.

Table 11. 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.

Of the eight programs releasing summer-run chum salmon that are considered to be part of the Hood Canal summer chum ESU, six of the programs are supplementation programs implemented to preserve and increase the abundance of native populations in their natal watersheds. 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 summer-run chum salmon.

Critical Habitat

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, which includes the Hamma Hamma and Dosewallips rivers and others, the Puget Sound subbasin, Dungeness/Elwha subbasin, and nearshore marine areas of Hood Canal and the Strait of Juan de Fuca from the line of extreme high tide to a depth of 30 meters. This includes a narrow nearshore zone from the extreme high-tide to mean lower low tide within several Navy security/restricted zones. This also includes about 8 miles of habitat that was unoccupied at the time of the designation Finch, Anderson and Chimacum creeks (69 FR 74572; 70 FR 52630), but has recently been re-seeded. Chimacum Creek, however, has been naturally recolonized since at least 2007 (T. Johnson, pers. comm., Jan. 2010). The designation for Hood Canal summer-run chum, like others made at this time, includes the stream channels within the designated stream reaches, and includes a lateral extent as defined by the ordinary high water line. In areas where the ordinary high-water line is not defined the lateral extent is defined as the bankfull 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 Hood Canal chum salmon ESU, 14 subbasins were rated as having a high conservation value, while only three were rated as having a medium value to 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 mainstem, excessive sediment in spawning gravels, and reduced stream flow in migration areas.

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 as much as adult fish spawn in the fall and winter, young emerge in the spring, rear in fresh water 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 three percent or less, which may be fed by cool groundwater sources, and are often widely dispersed within watershed. Adult coho salmon may remain in fresh water three or more months before spawning, with early migrants often moving farther upstream than later migrants (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 and other Pacific salmon are unable to enter the rivers until sufficiently strong flows open passages and provide sufficient depth. First fall freshets combined with high tides triggers 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.

Rates of incubation are largely temperature dependent: colder water temperatures will slow the rate of development. Generally, in optimal temperatures eggs incubate for about 35 to 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 fresh water for about 15 to 18 months. As fry grow, they disperse upstream and downstream to establish and defend territories. Juvenile rearing usually occurs in tributaries with gradients of three percent or less, although they may move to streams with gradients of four to five percent. 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, like other anadromous fish, undergo the physiological transition to salt water. The outmigration of coho 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). 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.

Coho salmon, like many other salmon, are opportunistic feeders. While at sea, coho salmon tend to eat fish including herring, sand lance, sticklebacks, sardines, shrimp and surf smelt. While in estuaries and in fresh water 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, plecoptera, and other larval insects, and typically use visual cues to find their prey.

Threats

Natural Threats. Coho salmon, like other salmon, are exposed to high rates of natural predation at each life stage. Most mortality, however, occurs in the freshwater life stages. Winter mortality may be significant for coho salmon because they overwinter in fresh water, 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).

Anthropogenic Threats. 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. These threats for are summarized in detail under Chinook salmon.

Central California Coast Coho Salmon

Distribution and Description of the Listed Species

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. 1995). 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. 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 Fisher Flats Conservation Program, Scott Creek Captive Broodstock Program, and the Noyo River Fish Station egg-take Program 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. 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. 2005).

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. 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. 2005).

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. 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. 2005).

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 and Description of the Listed Species

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, Twenty-five artificial propagation programs are part of this ESU: Grays River, Sea Resources Hatchery, Peterson Coho Project, Big Creek Hatchery, Astoria High School (STEP) Coho Program, Warrenton High School (STEP) Coho Program, Elochoman Type-S Coho Program, Elochoman Type-N Coho Program, Cathlamet High School FFA Type-N Coho Program, Cowlitz Type-N Coho Program in the Upper and Lower Cowlitz Rivers, Cowlitz Game and Anglers Coho Program, Friends of the Cowlitz Coho Program, North Fork Toutle River Hatchery, Kalama River Type-N Coho Program, Kalama River Type-S Coho Program, Lewis River Type-N Coho Program, Lewis River Type-S Coho Program, Fish First Wild Coho Program, Fish First Type-N Coho Program, Syverson Project Type-N Coho Program, Washougal River Type-N Coho Program, Eagle Creek NFH, Sandy Hatchery, and the Bonneville/Cascade/ Oxbow complex coho hatchery programs.

Two distinct runs distinguished by the timing of adult returns to fresh water (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 fresh water 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 adult coho salmon exhibit a northern oceanic distribution, return 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

NMFS listed Lower Columbia River coho salmon as endangered on June 28, 2005 (70 FR 37160). 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 12 , where available.

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. Most of the other populations are believed to have very little, if any, natural production. 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, despite the reductions in harvest.

Table 12. 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 totoal 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.

The most serious threat facing this ESU is the scarcity of naturally-produced spawners, with attendant risks associated with small populations, 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

NMFS has not designated critical habitat for Lower Columbia River coho salmon.

Southern Oregon/Northern California Coast Coho Salmon

Distribution and Description of the Listed Species

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. 1995). 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. 1995). 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, 1997). 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. 2005). 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. 2005). 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.

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.

Oregon Coast Coho Salmon

Distribution and Description of the Listed Species

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 13 identifies populations within the Oregon Coast coho salmon ESU, their abundances, and hatchery input.

Table 13. 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 to 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 to 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

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.

Sturgeon

Description of the Genus

Sturgeon (*Acipenseridae*) are one of the oldest *Osteichthyes* (bony fish) in existence, and are native to rivers and coastal areas of North America. The two listed sturgeon, discussed below, are part of the genus *Acipenser*, and share some common characteristics. Sturgeon, in general, have a characteristic external morphology distinguished by the inferior mouth typical of bottom-feeders. Most species are anadromous, although a few species are entirely fresh water and many species can survive if they become land-locked. Both listed species (discussed below) are anadromous and tend to remain in coastal waters. As an anadromous fish, sturgeon spawn in fresh water and feed and rear in marine or estuarine waters. Sturgeon are also iteroparous spawners and tend to be very long-lived.

Threats

Natural Threats. Freshwater predation of eggs and larvae from birds and larger fish, and marine predation of adult and subadult fish by sharks, pinnipeds and other large body predators.

Anthropogenic Threats. In general sturgeon have declined from the combined effects from the construction of dam and water diversion projects, dredging and blasting, water pollution, and fisheries. As a result of their longevity, slow rate of growth and delayed maturation, and bottom-feeding habits, in general sturgeon have a life history that makes them susceptible to over-harvest and exposure to (and the accumulation of) contaminants. Many sturgeon also do not spawn on an annual basis, but may spawn every other year or even more infrequently. Thus even small increases in mortality can affect population productivity (Heppell 2007). The body form and feeding habits of sturgeon may expose them to a different suite of contaminants or contaminant properties than pelagic fish due to their affinity with bottom sediments. Exposure pathways would include a dissolved or water borne exposure, but for sediment-associated contaminants the sediment exposure pathway may be more significant. Benthic dwelling fish may be exposed through the direct contact with sediment, exposed to the boundary layer over the sediment, and commonly have a higher rate of incidental ingestion and exposure through direct consumption of sediments.

Southern Green Sturgeon

Distribution and Description of the Listed Species

Green sturgeon occur along the west coast of North America from Mexico to the Bering Sea (Adams et al. 2002; Moyle 2002; Colway and Stevenson 2007). Distinguished primarily according genetic differences and spawning locations, NMFS recognizes two distinct population segments (DPS) of green sturgeon: a northern DPS whose populations are relatively healthy, and a Southern DPS that has undergone significant decline (Adams et al. 2007). NMFS listed the Southern DPS of green sturgeon as threatened in 2006 (71 FR 17757).

Green sturgeon are considered one of the most marine-oriented sturgeon species, spending much of their lives in coastal marine waters, estuaries and bays. Early life stages rear in fresh water, and adults return to fresh water when they are 15 years old or older to spawn. Across the species' range only three rivers contain documented spawning and only one of the rivers is part of the southern green sturgeon DPS, the Sacramento River (Moyle et al. 1992; CDFG 2002). Outside of natal rivers, the distribution of southern green sturgeon and northern green sturgeon overlap. 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.

Green sturgeon are spring spawners and initiate spawning migrations as early as March, spawn late spring to early summer, hold in deep pools and return to salt water in the fall early, often with the first increases in fall flows. There may a be a latitudinal cline in the timing of upstream spawning migrations, as fish in the Klamath River have been observed initiating migrations between April and June, Rogue River fish between May and July, whereas Heubein et al. (2009) observed Sacramento River fish making their upstream migrations between March and April. Spawning generally occurs in deep pools of large rivers or off-channel coves (Moyle et al. 1992, 1995; Erickson and Webb 2007; Erickson et al. 2001; Heublein et al. 2009; Rien et al. 2001). 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 (Heubein 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; Heubein 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.

Green sturgeon are a long-lived fish, and likely live for 60 to 70 years (Moyle 2002). Age at first maturation for green sturgeon is at least 15 years old, after which adults likely return every 2 to 5 years to spawn (Adams et al. 2002; Moyle 2002; 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 1973; Moyle et al. 1992). Temperature may trigger spawning behavior, with ranges of 48º to 62º F being influential (Moyle et al. 1995). Water temperature is also critical for egg survival with temperatures above 68º F being fatal to developing embryos (Cech et al. 2000).

Green sturgeon spend their first 1 to 4 years in their natal streams and rivers (Nakamoto et al. 1995; Beamsesderfer 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 et al. 2009a, b). Larvae are active at night, a behavior that likely reduces predation and avoids being moved downstream more than necessary (Cech et al. 2000). Green sturgeon larvae grow very rapidly, reaching about 300 mm by age one (Deng 2000). Temperature is strongly correlated with growth rates, with optimal growth rates occurring at about 59º F (Cech et al 2000). While in fresh water, juveniles feed on a variety of fishes and invertebrates (Moyle et al. 1992). 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).

Upon outmigration from fresh water, subadult green sturgeon disperse widely along through continental shelf waters of the west coast within the 110 meter contour (Moyle et al. 1992; NMFS 2005b; Erikson and Hightower 2007). Biologists have recaptured fish tagged in the Sacramento River, in coastal and estuarine waters to the north. It appears that green sturgeon generally distribute north of the river mouth from whence they emerge as juveniles during fall and move into bays and estuaries during summer and fall (Israel et al. 2009; Moser and Lindley 2007). The limited feeding data available for subadult and adult green sturgeon show that they consume benthic invertebrates including shrimp, clams, chironomids, copepods, mollusks, amphipods, and small fish (Houston 1988; Moyle et al. 1992; 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 2005b). 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 2005b). 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.

Threats

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 2007; 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 (Erickson et al. 2001; Moyle et al. 1992; Moyle et al. 1995; Rien et al. 2001). 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 2007). 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 2007). 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 2007). 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 2007). 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.

Shortnose Sturgeon

Distribution and Description of the Listed 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 (1998a) recognized 19 wild populations based on their strong fidelity to their natal streams, and several captive populations (from a Savannah River broodstock) that are maintained for educational and research purposes (NMFS 1998a; Table 14).

Shortnose sturgeon are generally anadromous (they migrate between sea and fresh water for reproductive purposes) or amphidromous (some fish migrate between fresh and salt water for reasons other than spawning, such as feeding), but such migratory behavior may not be obligatory for the species as they can also maintain land-locked (freshwater resident) populations. In general, shortnose sturgeon are benthic fish that occupy the deep channel sections of large rivers or estuarine waters of their natal rivers, and will migrate considerable distances. Dadswell (1979 in Dadswell et al. 1984) observed shortnose sturgeon traveling up 160 km between tagging and recapture in the St. John estuary, and it is not uncommon for adults to migrate 200 km or more to reach spawning areas (Kynard 1997).

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 (Northcote 1978; Harden-Jones 1968 in Dadswell 1984). 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 (see Dadswell et al. 1984 for a review).

Young-of-the year shortnose sturgeon move downstream after hatching, remaining in fresh water for about 1 year (Kynard 1997). Initially, young shortnose sturgeon will reside short distances from spawning areas, and as they grow will tend to move further downstream (Dadswell et al.

1984). 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).

Habitat use in fresh water during summer and winter months overlaps between adult and age-1 shortnose sturgeon (O'Herron et al. 1993; Rogers and Weber 1995 in Moser et al. 2000; 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, Rogers and Weber 1995, and Weber 1996 cited in Moser et al. 2000). 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.

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 (Dovel et al. 1992; Collins and Smith 1993; Kieffer and Kynard 1993; NMFS 1998a). 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. 1984; NMFS 1998a; Kynard et al., in draft).

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. 1984; 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. 1984; 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 1989). 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 1989 using data presented in Dadswell et al. 1984). Male shortnose sturgeon appear to have shorter life spans than females (Gilbert 1989).

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. 1984; 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. 1984). Based on observations by Kynard et al.

(2000), shortnose sturgeon will consume the entire mollusk, excreting the shell after ingestion.

Table 14. 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 1998a
	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 1998a
	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
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.

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. Although the original listing notice did not cite reasons for listing the species, a 1973 Resource Publication issued by the U.S. Department of Interior (USDOI), stated that shortnose sturgeon were "in peril ... gone in most of the rivers of its former range [but] probably not as yet extinct" (USDOI 1973). 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 its range

Northern shortnose sturgeon population abundances are generally larger than southern populations (Kynard 1997). 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 1998a; 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 (2003) 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. Based on studies from other sturgeon species we know of no evidence of senescence in sturgeon, and we would expect that these fish are reproductively active (Paramian et al. 2005). 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, and s. 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 species’ 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. 2005; and T. King, in progress)..

Threats

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 & 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 have also plagued Chesapeake Bay. In most cases, dissolved oxygen levels have improved 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 than 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 2007). Although sturgeon can spawn over varied benthic habitat, they prefer localized depressions in riverbeds (Erickson et al. 2001; Moyle et al. 1992; Moyle et al. 1995; Rien et al. 2001). 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 2007). 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 directly sensitive to elevated water temperatures. Temperature triggers spawning behavior. Warmer water temperatures can initiate spawning earlier in a season for salmon and the same can be true for sturgeon (ISAB 2007). 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 2007). 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 2007). 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

NMFS has not designated critical habitat for shortnose sturgeon.

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 throughout their range, 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 fresh water 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.

Once smolts enter the Pacific Ocean, they distribute widely across the North Pacific, generally above 40ºN 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 stay within several dozen feet of the surface, although they tend to be closer to the surface at night versus daytime (Manzer 1964; French et al. 1976). However, they may migrate several thousand miles in search of prey and are considered to travel continuously (Royce et al. 1968). 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). 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 (39 to 50º F) than larger, older fish (37 to 41º F), possibly an artifact of older fish being distributed further north. Adult upstream migration may be blocked by temperatures above 70º F (McCullough 1999). However, temperatures below 70º F can stress fish by increasing their susceptibility to disease and elevating their metabolism (Brett 1979; Berman 1990). 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 (Nishiyama 1984).

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 to 2,400 eggs per female 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 to less than 2,000 eggs.

Incubation is a function of water temperatures, but generally lasts between 100 and roughly 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).

Ozette Lake Sockeye Salmon

Distribution and Description of the Listed Species

This ESU includes all naturally spawned sockeye salmon in Ozette Lake, Ozette River, Coal Creek, and other tributaries flowing into Ozette Lake, Washington. 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 2009).

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 (Haggerty et al. 2009 in NMFS 2009). 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 fresh water.

Status and Trends

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 Umbrella Creek and Big River sockeye salmon hatchery programs. 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 *in* Good et al. 2005) 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 1998b). 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 2009). 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. 2005). 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. 2008 in NMFS 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. 2005). 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, 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 include: 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.

Snake River Sockeye Salmon

Distribution and Description of the Listed Species

Snake River sockeye salmon are unique compared to other sockeye salmon populations: it spawns at a higher elevation (6,500 feet) and a longer freshwater migration (approximately 900 miles) than any other sockeye salmon population in the world. 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.

Status and Trends

Snake River sockeye salmon were originally listed as endangered in 1991 and 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, in the Stanley Basin (upper Salmon River drainage) of Idaho. 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 fresh water, but spawn at the same time and in the same location as anadromous sockeye salmon. In 1993, 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).

NMFS has determined that this artificially propagated population is genetically no more than moderately divergent from the natural population (NMFS 2005a). 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. 2005). Only 18 natural origin sockeye salmon have returned to the Stanley Basin since 1987. During the fall of 1990, during the course of NMFS' first status review on the species, 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. 2005).

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.

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 1976; Stolz & Schnell 1991; NMFS 1997b). The life history of this species varies considerably throughout its range. Generally, steelhead can into two races: the stream-maturing type, summer steelhead, enters fresh water in a sexually immature condition and requires several months in fresh water to mature and spawn; and the ocean-maturing type, winter steelhead, enters fresh water 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 race.

Summer steelhead enter fresh water between May and October in the Pacific Northwest (Nickelson et al. 1992; Busby et al. 1996). 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 fresh water between November and April in the Pacific Northwest (Nickelson et al. 1992; Busby et al. 1996), 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).

There is a high degree of overlap in spawn timing between populations regardless of run type (Busby et al. 1996). 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, although steelhead rarely spawn more than twice before dying; most that do spawn more than twice tend to be female (Nickelson et al. 1992; Busby et al. 1996). Second time spawners often make up about 70 to 85 % of repeat spawners, with third time spawners make up between 10 to 25 % of repeats (Stolz & Schnell 1991). Iteroparity is more common among southern steelhead populations than northern populations (Busby et al. 1996).

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 fresh water 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 fresh water (Busby et al. 1996). 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 & Schnell 1991). 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 & Schnell 1991).

Threats

Natural Threats. Steelhead, like other salmon, are exposed to high rates of natural predation each stage of their 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 & Schnell 1991). In fresh water, fry fall prey to older steelhead and other trout, as well as birds, sculpin, and various mammals. In the ocean, marine mammals, and other fish prey on steelhead but the extent of such predation is not well known.

Anthropogenic Threats. 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. These threats for are summarized in detail under Chinook salmon.

Central California Coast Steelhead

Distribution and Description of the Listed Species

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. 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: the Don Clausen Fish Hatchery, and Kingfisher Flat Hatchery/ Scott Creek (Monterey Bay Salmon and Trout Project) steelhead hatchery programs.

The DPS is entirely composed of winter run fish, as are those DPSs to the south. As winter-run fish adults migrating upstream from December-April, and smolts emigrating 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 fresh water 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); threatened status was reaffirmed on January 5, 2006 (71 FR 834). Table 15 identifies runs within the Central California Coast steelhead DPS and their estimated run sizes.

Table 15. 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. 2005 for a discussion). Other estimates of historical abundance are on a per river basis: According to Busby et al. (1996), 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 (see Table 15). 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 1970 levels.

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 (see Good et al. 2005). 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. 2005 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. 2005). 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 in a way 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. 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 (fresh water 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 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 46 occupied watersheds reviewed in NMFS' assessment of critical habitat for Central California Coast steelhead, 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 and Description of the Listed Species

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. 1996; Zimmerman et al. 2009). 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 1996). The California Advisory Committee on Salmon and Steelhead (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 1996; 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 1996). Recent snorkel surveys (1999 to 2002) indicate that steelhead are present in Clear Creek (J. Newton, FWS, pers. comm. 2002, *in* Good et al. 2005). 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 void 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 et al. 2000). It is possible that naturally spawning populations exist in many other streams but are undetected due to lack of monitoring programs.

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 (CDFG 1995 in Good et al. 2005; McEwan and Jackson 1996). Steelhead in these basins travel extensive distances in fresh water (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 in from July through May, with peaks in September and February. Spawning begins in late December and can extend into April (McEwan and Jackson 1996).

Status and Trends

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 Central Valley steelhead 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 1996; McEwan 2001). The five-year geometric mean, however, is just under 2,000 steelhead (Table 16), and the long-term trend suggests that the population is declining.

Table 16. 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. 2005). In 2004, a total of 12 steelhead smolts were collected at Mossdale (CDFG, unpublished data *in* Good et al. 2005).

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 in Central Valley steelhead populations 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 1996). 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

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. 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 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 67 watersheds reviewed in NMFS' assessment of critical habitat for California Central Valley steelhead, 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 and Description of the Listed Species

Lower Columbia River steelhead include naturally produced steelhead returning to Columbia River tributaries on the Washington side between the Cowlitz and Wind rivers in Washington and on the Oregon side between the Willamette and Hood rivers, inclusive. In the Willamette River, the upstream boundary of this species is at Willamette Falls. This species includes both winter and summer steelhead. Two hatchery populations are included in this species, the Cowlitz Trout Hatchery winter-run population and the Clackamas River population but neither was listed as threatened. Table 17 identifies the populations that comprise Lower Columbia River steelhead and summarizes several measures available to characterize population viability.

Summer steelhead return sexually immature to the Columbia River from May to November, and spend several months in fresh water prior to spawning. Winter steelhead enter fresh water 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

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 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. 2005). 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 exceeded 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).

Table 17. 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				
	Tilton River		2,787[c]	73	
	Upper Cowlitz River				
	Lower Cowlitz River	1,672			
	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.

Critical Habitat

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. 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 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. Of the 32 watersheds reviewed in NMFS' assessment of critical habitat for Lower Columbia River steelhead, 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.

Middle Columbia River Steelhead

Distribution and Description of the Listed Species

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). Steelhead from the Snake River Basin (described elsewhere) are excluded from this DPS. Seven artificial propagation program are part of this DPS: The Touchet River endemic, Yakima River kelt reconditioning program (in Satus Creek, Toppenish Creek, Naches River, and the Upper Yakima River), and the Umatilla River and the Deschutes River steelhead hatchery programs. 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.

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 15.7 inches 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. 1996). According to Interior Columbia Basin Technical Recovery Team (ICBTRT 2003) 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). See Table 18 for a list of 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 18. 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]		1.011
		Lower NF - 3.52 rpm (1.5-8.8)		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	Yakima River			

Creek				
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. 1985). 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 the decline of Middle Columbia river steelhead 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. (1996) 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. 1996). Survey data collected between 1997 and 2001 indicates that several populations within the DPS have increased since the last status review (Good et al. 2005). However, long-term annual population growth rate (λ) is negative for most populations (see Table 18).

In contrast, short term trends in major areas were positive for 7 of the 12 areas with available data (see Good et al. 2005). 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. 2005). 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 in Good et al. 2005). Based on recent 5-year geometric means, only the Deschutes River exceeded interim target levels (Good et al. 2005). 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.

Critical Habitat

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. 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 bankfull 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.

Northern California Steelhead

Distribution and Description of the Listed Species

The Northern California DPS of steelhead includes all naturally spawned steelhead populations below natural and manmade impassible barriers in California coastal river basins from Redwood Creek south to, but not including the Russian river, and two artificial propagation programs (Yager Creek Hatchery, and North Fork Gualala River Hatchery). In the recent update on the status of this DPS, the southern boundary of the DPS was redefined to include the small coastal streams south of the Gualala River (between the Gualala River and the Russian River) that support steelhead. This DPS consists of winter and summer-run fish, as well as “half-pounders” – a sexually steelhead that returns from the sea after spending less than a year in the ocean. Generally, a half-pounder will overwinter in freshwater and return to the ocean in the spring.

Status and Trends

NMFS listed Northern California steelhead as threatened on June 7, 2000 (65 FR 36074), and reaffirmed their status as threatened on January 5, 2006 (71 FR 834). Long-term data sets are limited for Northern California steelhead. Before 1960, estimates of abundance specific to this DPS were available from dam counts in the upper Eel River (Cape Horn Dam; annual average number of adults was 4,400 in the 1940s), the South Fork Eel River (Benbow Dam; annual average number of adults was 18,000 in the 1940s), and the Mad River (Sweasey Dam; annual average number of adults was 3,800 in the 1940s). According to California Department of Fish & Game nearly 200,000 spawning steelhead may have comprised this DPS in the early 1960s (Good et al. 2005). At the time of the first status review on this population, adult escapement trends could be calculated for 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, there was and continues to be insufficient information to calculate an overall abundance estimate for Northern California steelhead (Busby et al. 1996).

Recent time series data is also limited for this DPS, with recent abundance estimates available for only four populations, three summer-run and one winter-run. Similarly, Good et al. (2005) could only calculate the population growth rate for three populations (see Table 19). Population growth rates are negative for two of the three populations, the South Fork Eel River winter-run and the Middle Fork Eel River summer-run. Based on time series data for the Middle Fork Eel River, both the long-term and short-term trends are downward. Due to the lack of adult data on which to base their risk assessment, Good et al. (2005) also examined data on juvenile steelhead, and found both upward and downward trends. The lack of data for the populations within this DPS, particular winter-run fish is of continuing concern.

Table 19. Northern California steelhead salmon populations and select measures of population viability

River	Historical Abundance[a]	Mean Number (CI)[b]	Growth Rate (λ)[c]
Redwood Creek	10,000	3 (n/a)	
Mad River	6,000	162 (162-384)[d]	1.00 (0.93,1.05)[e]
Freshwater Creek winter run		32 (25-32)	
Eel River -Total	82,000		
South Fork Eel River	34,000		0.98 (0.92,1.02)
Middle Fork Eel River	23,000	418 (384-1,246)[e]	0.98 (0.93,1.04)[g]
Mattole River	12,000		
Ten Mile River	9,000		
Noyo River	8,000		
Big River	12,000		
Navarro River	16,000		
Garcia River	4,000		
Gualala River	16,000		
Other Humboldt County streams	3,000		
Other Mendocino County streams	20,000		

[a]Historical abundances (1963) are considered uncertain by the author, California Department of Fish & Game. All data are reported in Good et al. 2005.
[b]Value represents the geometric mean number of fish surveyed by snorkel counts or weir counts (e.g., Mad River and MF Eel counts are from snorkel surveys – for the MF Eel River these are snorkel counts of fish holding in pools of the main stem). See Good et al. 2005 for details.
[c]Growth rate calculated upon method where a λ=1.0 could describe a population that is in decline due to environmental stochasticity.

[d]Five year mean of Mad River summer-run steelhead only.
[e]Population growth rate calculated on Mad River winter-run steelhead only.

Critical Habitat

NMFS designated critical habitat for Northern California steelhead 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, and the Mendocino Coast. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. 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 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, Northern California steelhead occupy 50 watersheds (fresh water and estuarine). The total area of habitat designated as critical includes about 3,000 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 50 watersheds reviewed in NMFS' assessment of critical habitat for Northern California steelhead, nine watersheds received a low rating of conservation value, 14 received a medium rating, and 27 received a high rating of conservation value for the species. Two estuarine areas used for rearing and migration (Humboldt Bay and the Eel River estuary) also received a rating of high conservation value.

Puget Sound Steelhead

Distribution and Description of the Listed Species

The Puget Sound DPS for steelhead includes all naturally spawned anadromous winter-run and summer-run steelhead populations in watersheds of the Strait of Juan de Fuca, Puget Sound and Hood Canal, Washington. Boundaries of this DPS extend to and include the Elwha River to the west, and the Nooksack River and Dakota Creek to the north. Hatchery production of steelhead is widespread throughout this DPS, but only two artificial propagation programs are part of this DPS: the Green River natural and Hamma Hamma winter-run steelhead hatchery populations. The remaining hatchery programs are not considered part of the Puget Sound steelhead DPS because they are more than moderately diverged from the local native populations (NMFS 2005c).

The oceanic distribution of Puget Sound steelhead is not well understood. Winter and summer runs from multiple DPS' comingle in the North Pacific Ocean and some may undergo extensive migrations as a result of the location of their natal streams and oceanic "centers of abundance" (Light et al. 1989). Tagging and genetic studies indicate that Puget Sound steelhead migrate to the central North Pacific ocean (see French et al. 1975, Hartt and Dell 1986, and Burgner et al. 1992 in NMFS 2005c). However, the fjord-like ecosystem of Puget Sound 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. Oceanic residence times varies among populations within the DPS, with some populations spending only one season in the ocean and others spending three years in marine waters before returning to their natal stream for spawning. Generally, winter-run steelhead enter their natal freshwater systems later (November to April) in the year than summer-run steelhead (May to October), and thus have a shorter freshwater residence time just prior to spawning. The result is that winter-run steelhead have a lower pre-spawn mortality rate than summer-run steelhead (NMFS 2005c). Winter-run steelhead are also more prevalent than summer-run fish, comprising 37 of the 53 populations within this DPS.

Status and Trends

NMFS listed Puget Sound steelhead as a threatened species on May 11, 2007 (72 FR 26722). At the time of the listing, the biological review team concluded that: the viability of Puget Sound steelhead is at a high risk due to declining productivity and abundance; Puget Sound steelhead are at moderate risk due to reduced spatial complexity and connectivity among populations within the DPS, and reduction in life-history diversity within populations and from the threats posed by artificial propagation and harvest. The Puget Sound steelhead DPS includes 53 putative populations; most of which are composed of winter-run fish. Summer-run populations within Puget Sound are small, with most averaging less than 200 spawners, and most lack sufficient data to estimate population abundance. Table 20 lists several of the populations that comprise Puget Sound steelhead as well as some statistics summarizing their current status.

In general, steelhead are most abundant in the northern Puget Sound streams. The largest populations in this DPS are in the Skagit River and Snohomish River winter-run steelhead populations. The recent geometric mean escapement is 5,608 winter-run steelhead in the Skagit, and 3,230 winter-run steelhead in the Snohomish River. The Green River and Puyallup River populations, in central Puget Sound, are the next largest populations and average approximately 1,500 (Green) and 1,000 (Puyallup) winter-run steelhead spawners annually.

Table 20. Puget Sound steelhead salmon populations and a summary of available demographic data

Population	Life History	Historical Abundance (Percent Annual change[a]	Mean Number of Spawners[b]	Trends in escapement[c]	Median short-term growth rate (λ)[d]
Canyon	Summer				
	Winter				
Skagit	Summer				

	Winter	7,700 (2.0)	5608.5	-0.002	0.997 (0.997-0.998)
Snohomish	Summer				
Snohomish	Winter	8,000 (3.1)	3230.1	-0.019	0.804 S
Dakota	Winter				
Nooksack	Winter	NA (-11.6)			
Samish	Winter		852.2	0.067**	0.988 (0.997-0.998)
Stillaguamish	Winter	NA (-6.3)	550.2	-0.065****	0.885 S (0.884-0.885)
Tolt	Summer		119.0	0.025	1.018 (1.017-1.018)
Green	Summer				
Green	Winter		1625.5	0.008	0.932 (0.932-0.933)
Cedar	Winter		36.8	-0.179**	0.808 S (0.804-0.811)
Lake Washington	Winter	NA (-17.5)	36.8	-0.180****	0.802 (0.800-0.803)
Nisqually	Winter	1,200 (-5.1)	392.4	-0.084****	0.918 (0.917-0.918)
Puyallup	Winter	2,000 (-5.2)	1001.0	-0.062****	0.882 (0.881-0.882)
Dewatto	Winter		24.7		1.020 (1.008-1.020)
Dosewallips	Winter		76.7		
Duckabush	Winter		17.7	0.017	
Hamma Hamma	Winter		51.9	0.291*	1.013
Quilcene	Winter		15.1	-0.006	0.988 S
Skokomish	Winter	NA (-3.5)	202.8	-0.075****	0.865 S
Tahuya	Winter	NA (-0.6)	117.0	0.009	0.983 (0.982-0.983)
Union	Winter		55.3	0.008	0.969 S
Elwha	Summer				
	Winter		210.0		0.966 (0.965-0.966)
Dungeness	Winter	NA (-5.5)	173.8	-0.076	0.924 (0.924-0.924)
Mc Donald	Winter			-0.031	0.732 S
Morse	Winter	200 (-12.3)		-0.006	0.945 (0.945-0.946)

[a]Values of historical abundance represent the total escapement for the subbasin. Data generally span the late 1970s to mid 1990s. All estimates are run reconstructions, except the Nooksack which comes from spawner surveys. Specific data years for each data set and other details are noted in Busby et al. 1996.
[b]Geometric mean estimates of escapement for Puget Sound steelhead are provided for the five year period from 2000-2004, and for hatchery plus natural spawners (NMFS 2005c).
[cr]Estimates of temporal trends in escapement and total run size (transformed by natural log). Estimates are the slopes of the regressions of natural log (spawners or run size) on year. Estimates provided are for the entire available dataset and are based on natural fish (data years noted in NMFS 2005c). *, P<0.05; **, P<0.01; ***, P<0.001; ****, P<0.0001 (all other values are not significant (data from NMFS 2005c)).
[d]Estimates for each population were computed for the most recent 10 years of data (1995-2004). S – Denotes that the estimate is based on natural spawners alone. Values in parentheses represent the 95% Confidence Intervals of the estimate (data from NMFS 2005c).

Estimates of historical abundance for this DPS are largely based on catch data. The earliest catch records from commercial fisheries in the late 1880s indicate that the catch peaked at 163,796 steelhead in Puget Sound in 1895 (NMFS 2005c). Based on this catch data, NMFS (2005c) estimated that the peak run size for Puget Sound steelhead ranged between 300,000 and 550,000 fish. Given that most fish were harvested in terminal fisheries (nets set at the mouth of rivers) NMFS expects that this estimate is a fair estimate of the Puget Sound DPS as it is unlikely to include fish from neighboring rivers outside of the Puget Sound DPS. As early as 1898, Washington officials expressed concerns that the run had declined by half of its size in only three years (NMFS 2005c). Since 1925, Washington has managed steelhead as a game fish, and in 1932 the State prohibited the commercial catch, possession or sale of steelhead.

Run size for this DPS 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. Escapement of wild fish to spawning grounds would be much lower without the influx of hatchery fish (Busby et al 1996).

NMFS first status review for Puget Sound steelhead demonstrated that 80 % of the runs for which there was data had declining trends in abundance. Basinwide abundance estimates from Busby et al. (1996) are depicted in Table 20. Busby et al. (1996) noted that the largest decline, an 18% annual decline, occurred in the Lake Washington population. On the contrary, the largest increase in abundance occurred in the Skykomish River winter-run steelhead (the Skykomish River is a tributary to the Snohomish River) at a 7% annual increase. Estimates of spawner abundance in the Skagit and Snohomish rivers, the two largest steelhead producing basins in the DPS, were about 8,000 naturally spawning adult steelhead each (Table 20). These two basins exhibited modest overall upward trends at the time of the first status review. Recent data demonstrates significant declines in the natural escapement of steelhead throughout the DPS, especially in the southern Puget Sound populations. Significant positive trends have occurred in the Samish and the Hamma Hamma winter-run populations. The increasing trend in the Hamma Hamma River appears to be the result of a captive rearing program, rather than due to natural escapement. The predominant downward trends in escapement and run size of natural steelhead in the Puget Sound DPS, both over the long-term and short-term, is of concern particularly given that despite widespread reductions in direct harvest since the mid 1990s (NMFS 2005c).

Critical Habitat

NMFS has not designated critical habitat for Puget Sound steelhead.

Snake River Steelhead

Distribution and Description of the Listed Species

The Snake River Basin steelhead DPS includes all naturally spawned populations of steelhead in streams in the Snake River basins of southeast Washington, northeast Oregon and Idaho. Six artificial propagation programs are considered part of this DPS: The Tucannon River, Dworshak National Fish Hatchery, Lolo Creek, North Fork Clearwater, East Fork Salmon River, and the Little Sheep Creek/Imnaha river hatchery programs. These artificially propagated populations are no more divergent relative to the local natural populations than what would be expected between closely related natural populations within the DPS.

Snake River Basin steelhead are distributed throughout the Snake River drainage basin, migrating a considerable distance from the ocean to use high-elevation tributaries (typically 1,000-2,000 m above sea live). Generally, classified as summer-run fish, Snake River steelhead enter the Columbia River from late June to October. After remaining in the river through the winter, Snake River steelhead spawn the following spring (March to May). Managers recognize two life history patterns within Snake River steelhead primarily based on ocean age and adult size upon return: A-run steelhead are typically smaller, have a shorter fresh water and ocean residence (generally 1 year in the ocean), and begin their up-river migration earlier in the year; whereas B-run steelhead are larger, spend more time in fresh water and the ocean (generally 2-years in ocean), and appear to start their upstream migration later in the year. Table 21 lists the life-history type associated with each of the 24 demographically independent populations within this DPS.

Table 21. Snake River steelhead populations and a summary of available demographic data

Populations[a]	Life History	Historical Abundance (Percent Annual change[b]	Mean Number of Spawners (range)[c]	Percent Hatchery Contribution[d]	Long-term growth rate (λ)[e]
Tucannon River	A-run	400 (-18.3)	407 (257-628)	74	0.886, 0.733
Asotin Creek	A-run	200 (-19.7)	87 exp. redds (0-543)	Unknown	
Lower Clearwater	A-run				
South Fork Clearwater	B-run				
Lolo Creek	B-run				
Selway River	B-run				
Lochsa River	B-run				
North Fork Clearwater River					
Lower Grande Ronde	A-run	(-0.5)			
Joseph Creek	A-run		1,542 (1.077-2,385)	0	1.069
Wallowa River	A-run	(-3.0)			
Upper Grande Ronde	A-run		1.54 rpm (0.3-4.7)	23	0.967, 0.951
Little Salmon and lower Salmon tributaries	A-run				
South Fork Salmon River	B-run	(-8.0)			
Secesh River	B-run				
Chamberlain Creek	A-run				
Lower Middle Fork Salmon	B-run	(-25.8**)			
Upper Middle Fork Salmon	B-run				
Panther Creek	A-run				
North Fork Salmon	A-run				
Lemhi River	A-run				
Pahsimeroi River	A-run	1,400 (0.1)			
East Fork Salmon River	A-run	150*(-6.0)			
Upper Mainstem Salmon River	A-run				
Imnaha River	A-run	(81.2)	3.7 rpm (2.0-6.8)	20	1.042, 1.026
Hells Canyon tributaries	A-run				

[a] Demographically independent populations identified by ICBTRT 2003.
[b]Values of historical abundance represent total escapement as calculated in NMFS' first status review for the DPS. Values with a * are estimates of total run; no escapement estimate was available. Data generally span the late 1980s to mid 1990s. Estimates are calculated from different data types, and include data from spawner surveys, run reconstructions, or dam/weir counts. Specific data years for each data set and other details are noted in Busby et al. 1996. **=Middle Fork and tributaries.
[c]Geometric mean estimates of escapement represent total escapement (hatchery plus natural adult returns).
[c]Estimates of percentage of hatchery returns in Granite dam aggregate counts indicate that returns are predominantly composed of hatchery fish (about 85%). Values from Good et al. 2005.
[c]Multiple estimates for long-term growth (λ) presented for some populations represent two different assumptions on the contribution of hatchery fish to 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. 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.

Status and Trends

NMFS listed Snake River steelhead as threatened in 1997 (62 FR 43937), and reaffirmed their status as threatened on January 5, 2006 (71 FR 834). NMFS 1997 status review identified sharp declines in the returns of naturally produced steelhead, beginning in the mid-1980s. At the time nine of 13 trend indicators were in decline and the average abundance (geometric mean, 1992-1996) for the DPS was 75,000 adult steelhead (8,900 naturally produced). Of this, about 7,000 were A-run adults, and about 1,400 were B-run adults (Busby et al. 1996).

The paucity of information on adult spawning escapement for specific tributaries of the Snake River Basin DPS continues to make a quantitative assessment of viability difficult. Available data indicate that the overall long-term estimates of population trends have remained negative. Return estimates for the late 1990s to early 2000s are summarized in Table 21. Annual return estimates are limited to counts of the aggregate return over Lower Granite Dam, and spawner estimates for the Tucannon, Asotin, 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 geometric 5-year mean abundance (Total escapement 106,175 with 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 is 28,303 adults. Long-term trend estimates of the population growth rate (λ) across the available data set was 0.998 assuming that natural returns are produced only from natural-origin spawners, and 0.733 if both hatchery and wild spawners are contributing to production equally. 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 populations occurs within basins and some populations exhibit high stray rates.

Critical Habitat

NMFS designated critical habitat for Snake River steelhead on September 2, 2005 (70 FR 52630). Designated critical habitat includes the following subbasins: Hells Canyon, Imnaha River, Lower Snake/Asotin, Upper Grand Ronde River, Wallowa River, Lower Grand Ronde, Lower Snake/Tucannon, Upper Salmon, Pahsimeroi, Middle Salmon-Panther, Lemhi, Upper Middle Fork Salmon, Lower Middle Fork Salmon, Middle Salmon, South Fork Salmon, Lower Salmon, Little Salmon, Upper and Lower Selway, Lochsa, Middle and South Fork Clearwater, and the Clearwater subbasins, and the Lower Snake/Columbia River corridor. These areas are important for the species’ overall conservation by protecting quality growth, reproduction, and feeding. 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.

There are 289 watersheds within the range of Snake River steelhead. The total area of habitat designated as critical includes about 8,000 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 bankfull elevation. Of the 289 fifth order streams reviewed in this DPS, 231 received a high conservation value rating, 44 received a medium rating, and 14 received a rating of low conservation value for the species. The lower Snake/Columbia rearing/migration corridor downstream of the spawning range has a high conservation value. Limiting factors identified for Snake River Basin steelhead include: hydrosystem mortality, reduced stream flow, altered channel morphology and floodplain, excessive sediment, degraded water quality, harvest impacts, and hatchery impacts.

South-Central California Coast Steelhead

Distribution and Description of the Listed Species

The South-Central California Coast 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. No artificially propagated steelhead populations that reside within the historical geographic range of this DPS are included in this designation. The two largest basins within this DPS are the inland basins of the Pajaro River and the Salinas River. Both of these watersheds drain intercoastal mountain ranges and have long alluvial lower stretches. Principle sub-basins in the Pajaro River that support steelhead include: Corralitos Creek, Pescadero Creek, Uvas Creek, and Pacheco Creek. Principle sub-basins in the Salinas River that support steelhead include the Arroyo Seco River, Gabilan Creek, Paso Robles Creek, Atascadero Creek and Santa Margarita Creek. Other important watersheds include the smaller coastal basins of the Carmel River, and St. Rosa and San Luis Obispos creeks.

Status and Trends

NMFS listed South-Central California Coast steelhead as threatened in 1997, and reaffirmed their status as threatened on January 5, 2006 (71 FR 834). Historical data on the South-Central California Coast steelhead DPS are sparse and no credible historic or recent estimates of total DPS size are available. Steelhead are present in a large portion of the historically occupied basins within this DPS (estimated 86-95 %) but observed and inferred abundance suggest many of this basins support a small fragment of their historic run size. Present population trends within individual watersheds continuing to support runs is generally unknown, but may vary widely between watersheds. No data are available to estimate the steelhead abundance or trends in the two largest watersheds in the DPS, the Pajaro and Salinas basins, although these basins are highly degraded and expected to support runs much reduced in size from historical levels.

Steelhead in the Carmel Basin have been monitored at San Clemente Dam since 1964, representing one of the longest data sets available for steelhead in this DPS. However, this data is also limited because a nine year gap exists in the series, a large portion of the run spawns below the dam, and the older dam counts may be incomplete. Between NMFS’ 1997 status review and 2005 status update, continuous data from San Clement dam suggests that the abundance of adult spawners in the Carmel River has increased. Carmel River time series data indicate that the population declined by about 22% per year between 1963 and 1993, and between 1991 and 1997 the population increased from one adult to 775 adults at San Clemente Dam. Good et al. (2005) deemed this increase 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. The five-year geometric mean calculated by Good et al. (2005) for the Carmel River population (1998-2002) was 611 steelhead (range 1-881).

Critical Habitat

NMFS designated critical habitat for South-Central California Coast steelhead on September 2, 2005 (70 FR 52488). Specific geographic areas designated include the following CALWATER hydrological units: Pajaro River, Carmel River, Santa Lucia, Salinas River, and Estero Bay. These areas are important for the species’ overall conservation by protecting quality growth, reproduction, and feeding. 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 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, South-Central California Coast steelhead occupy 30 watersheds (fresh water and estuarine). The total area of habitat designated as critical includes about 1,250 miles of stream habitat and about 3 square miles of estuarine habitat (e.g., Morro 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 30 watersheds reviewed in NMFS' assessment of critical habitat for South-Central California Coast steelhead, six watersheds received a low rating of conservation value, 11 received a medium rating, and 13 received a high rating of conservation value for the species.

Southern California Steelhead

Distribution and Description of the Listed Species

The Southern California steelhead DPS includes all naturally spawned populations of steelhead in streams from the Santa Maria River, San Luis Obispo County, California (inclusive) to the United States-Mexico border. Artificially propagated steelhead that reside within the historical geographic range of this DPS are not included in the listing.

A comprehensive assessment of the distribution of steelhead within the Southern California DPS indicates that steelhead occur in most of the coastal basins (Boughton and Fish 2003 in Good et al. 2005). Major watersheds occupied by steelhead in this DPS include the Santa Maria, Santa Ynez, Ventura, Santa Clara rivers. Smaller watersheds that support steelhead include the Los Angeles, San Gabriel, San Luis Rey, and Sweetwater rivers, and San Juan and San Mateo creeks. Significant portions of several upper watersheds are contained with four national forests (Los Padres, Angeles, Cleveland, and San Bernardino National Forests), whereas coastal and inland valleys are dominated by urban development, with the Los Angeles basin being the most expansive and densest urban area in the DPS. Populations within the southernmost portion of the DPS (San Juan Creek, San Luis Rey River, and San Mateo Creek) are separated from the northernmost populations by about 80 miles.

Status and Trends

NMFS listed Southern California steelhead as endangered in 1997 (62 FR 43937), and reaffirmed their status as endangered on January 5, 2006 (71 FR 834). Historical and recent data is generally lacking for Southern California steelhead, making a general assessment of their status difficult. The historical run size estimate for the entire DPS was between 32,000-46,000 steelhead, but this estimate omits the Santa Maria system and basins south of Malibu Creek (Busby et al. 1996). Estimates for the Santa Ynez River Basin, probably the largest run historically, range from 13,000 to 30,000 spawners, although this number may underestimate the steelhead abundance in the basin prior to the construction of Juncal and Gibraltar dams (Busby et al. 1996; Good et al. 2005). No recent data are available for steelhead in the Santa Ynez basin, and most of the historical spawning habitat was blocked by Bradbury and Gibraltar dams. Steelhead and rainbow trout are known to occur in streams downstream of Bradbury Dam, but no estimates of abundance or trends are available. Similarly, Twitchell Dam in the Santa Maria River, and Casitas Dam on Coyote Creek and Matilija Dam on Matilija Creek block access to significant portions of historical spawning and rearing habitat, and alter the hydrology of the basins. A fish ladder and counting trap at the Vern Freeman Diversion Dam on the Santa Clara River is thought to be dysfunctional (Good et al. 2005). In general run sizes in river systems within the DPS are believed to range between less than five anadromous adults per year, to less than 100 anadromous adults per year. An estimated 26-52% of historically occupied basins are believed to still contain some steelhead, and about 30% are believed vacant, extirpated or nearly extirpated due to dewatering or barriers that block spawning habitat.

Critical Habitat

NMFS designated critical habitat for Southern California steelhead on September 2, 2005 (70 FR 52488). Specific geographic areas designated include the following CALWATER hydrological units: Santa Maria River, Santa Ynez, South Coast, Ventura River, Santa Clara Calleguas, Santa Monica Bay, Callequas, and San Juan hydrological units. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. 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 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, Southern California steelhead occupy 32 watersheds (fresh water and estuarine). The total area of habitat designated as critical includes about 700 miles of stream habitat and about 22 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 32 watersheds reviewed in NMFS' assessment of critical habitat for Southern California steelhead, five watersheds received a low rating of conservation value, six received a medium rating, and 21 received a high rating of conservation value for the species.

Upper Columbia River Steelhead

Distribution and Description of the Listed Species

The Upper Columbia River steelhead DPS includes all naturally spawned populations of steelhead in streams in the Columbia River Basin upstream from the Yakima River, Washington, to the United States-Canada border. Six artificial propagation programs are part of this DPS: the Wenatchee River, Wells Hatchery (in the Methow and Okanogan rivers), Winthrop National Fish Hatchery, Omak Creek, and the Ringold steelhead 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 DPS.

Rivers in this DPS primarily drain the east slope of the northern Cascade Mountains and include the Wenatchee, Entiat, Methow, and Okanogan River Basins. Some of these upper Columbia River subbasins, including the Okanogan River and the upper Columbia River proper, extend into British Columbia although steelhead do not occur in significant numbers in British Columbia and thus were not included in the DPS. Identified largely on the basis of spawning distributions, this DPS is composed of four putative populations defined by the Wenatchee, Entiat, Methow, and Okanogan rivers (Table 22). Historically (before the construction of Grand Coulee Dam blocked 50% of the river to Upper Columbia steelhead) major watershed that may have supported steelhead within this DPS were the Sanpoil, Spokane, Colville, Kettle, Pend Oreille and Kootenai rivers (ICBTRT 2003).

All upper Columbia River steelhead are summer-run steelhead. Adults return in the late summer and early fall, with most migrating relatively quickly to their natal tributaries. A portion of the returning adult steelhead overwinters in mainstem reservoirs, passing over upper-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 1 to 7 years rearing in fresh water before migrating to sea. Smolt outmigrations are predominantly year class two and three (juveniles), although some of the oldest smolts are reported from this DPS (7 years). Most adult steelhead return to fresh water from sea after 1 or 2 years.

Status and Trends

NMFS originally listed Upper Columbia River steelhead as endangered in 1997 (62 FR 43937). On January 5, 2006, after reviewing the status of Upper Columbia River steelhead and noting an increase in abundance and more widespread spawning, NMFS reclassified the status of Upper Columbia River threatened (71 FR 834). In accordance with a United States District Court decision, NMFS reinstated the endangered status of Upper Columbia River steelhead in June 2007 (62 FR 43937). NMFS appealed the Court's decision, and on June 18, 2009, the District Court revised its ruling, effectively reinstating threatened status for Upper Columbia River steelhead (74 FR 42605). Thus, consistent with the court's rulings and the NMFS' listing determination of January 5, 2006, Upper Columbia River steelhead are listed as threatened under the ESA.

Since the 1940s, artificially propagated steelhead have seeded this DPS to supplement the numbers lost with the construction Grand Coulee Dam. 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). Early estimates of steelhead in this DPS may be based on runs that were already depressed due to dams and steelhead fisheries. Nevertheless, these early dam counts are the best source of available data on the former size of the populations within this DPS. From 1933-1959 counts at Rock Island Dam averaged between 2,600 and 3,700 steelhead adults, which suggested the pre-fishery run size likely exceeded 5,000 adults destined for tributaries above Rock Island Dam (Chapman et al. 1994 in Busby et al. 1996). Using counts at Priest Rapids Dam (located below the production areas for this DPS) as an indicator of DPS size and trends suggests that the total number of spawners has increased since NMFS' 1996 status review. The 1992-1996 average annual total returns (hatchery plus natural) of steelhead spawners was 7,800, and the 1997-2001 average is 12,900 steelhead (hatchery plus natural). The natural component increased in these same periods from 1,040 to 2,200, respectively (Good et al. 2005).

Table 22. Upper Columbia River steelhead salmon populations and a summary of demographic data

Population	**Historical Abundance (Percent Annual change)**[a]	**Mean Number of Spawners (range)**[b]	Percent Hatchery Contribution[c]	Long-term growth rate (λ)[d]
Wenatchee River	2,500 (2.6)	3,279** (1,899-8,036)	71 (65)	1.067, 0.733
Entiat River				

Methow River Okanogan River	2,400* (-12.0)	3,714** (1,879-12,801)	91 (81)	1.086, 0.589

[a]Values of historical abundance represent total escapement as calculated in NMFS' first status review for the DPS. * = value represents a combined total escapement for the Methow and Okanogan rivers. Available data series: Wenatchee = 1962-1993, Methow and Okanogan = 1982-1993; calculations represent the geometric mean 1989-1993. Estimates are run reconstructions. Demographically independent populations identified by ICBTRT 2003.
[b]Geometric mean estimates of escapement represent total escapement (hatchery plus natural adult returns). ** Estimates of the mean number of spawners is a combined estimate for the Wenatchee and Entiat rivers, and the Methow and Okanogan rivers are also combined.
[c]Estimates of percentage of hatchery returns are from Good et al. 2005, and are based on extrapolations from mainstem dam counts and sampling. Parenthetical values are from Busby et al. 1996, and are provided for comparison.
[d]Multiple estimates for long-term growth (λ) are provided by Good et al. (2005) and represent two different assumptions on the contribution of hatchery fish to natural production. The first value reflects the assumption that hatchery fish do not contribute to natural production, and the second value reflects the assumption that hatchery fish contribute to natural production at the same rate as natural-origin spawners. Data series: 1976-2001.

While the total number of naturally produced fish in this DPS increased between status reviews, the proportion of naturally produced steelhead to hatchery-origin fish has declined. Total escapement increased in the combined estimate for the Wenatchee and Entiat rivers to a geometric mean of 3,279 spawners (900 natural spawners) over NMFS' previous estimate of 2,500 hatchery and natural steelhead spawners (1989 to 1993, natural component 800 steelhead). Estimates of the hatchery contribution to this population increased from 65% to 71% of total escapement (Table 22). A comparison of estimates for the Methow and Okanogan rivers during the same periods indicate that the total escapement increased from 2,400 to 3,714 while naturally produced steelhead declined from 450 to 358. Thus, the contribution of naturally produced steelhead declined from 19% to only 9% of total escapement between the 1993 and 2001 estimates (Good et al. 2005).

The assumptions of the role that hatchery fish play in the overall productivity and health of the DPS strongly influence estimates of population growth rates. Estimates based on the assumption that hatchery fish contribute to natural production at the same rate as natural-origin spawners consistently result in long-term population growth rates (expressed as λ) that are consistently below 1 (Table 22). Under the assumption that hatchery fish do not contribute to natural production, estimates of long term population growth rate suggest the population is growing. Determining the actual contribution of hatchery fish to natural production is important for understanding the true status of this DPS, particularly given that the proportion of naturally produced steelhead to hatchery-origin steelhead continues to decline. The extremely low replacement rate of naturally produced steelhead in this DPS is of concern, and the returns of natural steelhead remain well below recovery target levels.

The majority of the biological review team (54%) felt that this DPS warranted an "endangered" listing due to the growth rate and productivity, and uncertainty over the contribution of hatchery fish to natural production. NMFS, after convening a review of the artificial propagation programs of the six hatcheries in the DPS concluded that the programs collectively mitigate the immediacy of extinction risk in the DPS. Thus, NMFS listed the DPS as threatened rather than threatened (71 FR 834). NMFS concluded that the hatchery programs have increased total escapement and the distribution of spawning areas, and minimize the potential risks associated with artificial propagation. However, the abundance and productivity of naturally spawned steelhead remains a concern.

Critical Habitat

NMFS designated critical habitat for Upper Columbia River steelhead on September 2, 2005 (70 FR 52630). Designated critical habitat includes the following subbasins: Chief Joseph, Okanogan, Similkameen, Methow, Upper Columbia/Entiat, Wenatchee, Lower Crab, 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. 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.

There are 42 watersheds within the range of Upper Columbia River steelhead. The total area of habitat designated as critical includes about 1,250 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 bankfull elevation. Of the 42 watersheds reviewed in NMFS' assessment of critical habitat for Upper Columbia River steelhead, three watersheds received a low rating of conservation value, eight received a medium rating, and 31 received a high rating of conservation value for the species. In addition, the Columbia River rearing/migration corridor downstream of the spawning range was rated as a high conservation value. Limiting factors identified for the Upper Columbia River steelhead include: mainstem Columbia River hydropower system mortality, reduced tributary stream flow, tributary riparian degradation and loss of in-river wood, altered tributary floodplain and channel morphology, and excessive fine sediment and degraded tributary water quality.

Upper Willamette River Steelhead

Distribution and Description of the Listed Species

The Upper Willamette River steelhead DPS includes all naturally spawned populations of winter-run steelhead in the Willamette River, Oregon, and its tributaries upstream from Willamette Falls to the Calapooia River (inclusive). No artificially propagated populations that reside within the historical geographic range of this DPS are included in this listing. Hatchery summer-run steelhead occur in the Willamette Basin but are an out-of-basin population that is not included in this DPS.

The native (late) winter-run steelhead, with spring Chinook salmon, are the only two populations of salmon believed to historically occur above Willametter Falls (RKm 77). The construction of a fish ladder at the falls in the late 1880s, allowed for the passage of summer steelhead from Skamania Creek and winter-run steelhead from Big Creek (i.e., Gnat Creek). The two groups of winter-run steelhead exhibit different return times. The later run exhibits the historical phenotype adapted to passing the seasonal barrier that existed at Willamette Falls prior to construction of the fish ladder. The early run of winter-run steelhead are considered non-native, and were derived from Columbia River steelhead outside the Willamette River (Good et al. 2005). While the release of these hatchery winter-run fish was recently discontinued, some fish from earlier releases now reproduce naturally within the upper Willamette River Basin. Nonnative summer-run hatchery steelhead continue to be released within the upper basin (Good et al. 2005).

Native steelhead in the Upper Willamette are a late-migrating winter group that enters fresh water in January and February (Howell et al. 1985). They do not ascend to their spawning areas until late March or April (Dimick and Merryfield 1945) and spawning occurs from April to June 1. 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. 1985). Smolts generally migrate through the Columbia via Multnomah Channel rather than the mouth of the Willamette River. Most spend 2 years in the ocean before re-entering natal rivers to spawn (Busby et al. 1996). 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. 1996).

Status and Trends

NMFS originally listed Upper Willamette River steelhead as threatened in 1999 (64 FR 14517), and reaffirmed their status as threatened on January 5, 2006 (71 FR 834). The Upper Willamette steelhead DPS consists of four demographically independent populations, each of which remains extant although depressed from historical levels (Table 23). Available data for this DPS comes from a combination of dam counts, redd count index surveys, and hatchery trap counts. Estimates of abundance from NMFS 1996 status review on this DPS, demonstrate a mix of trends with the largest populations, Mollala and North Santiam rivers, declining over the period of analysis. The 2005 review of the status of the Upper Willamette steelhead DPS indicated that each population showed a declining trend over the data series that extended to 2000 and 2001, while one population, the Calapooia River, increased over the short-term (1990-2000/1; Good et al. 2005).

More recently, data reported in McElhany et al. (2007) indicate that currently the two largest populations within the DPS are the Santiam River populations. Mean spawner abundance in both the North Santiam River and the South Santiam River is about 2,100 native winter-run steelhead. Long-term growth is negative for three of the populations within the DPS, with Calapooia River demonstrating a lambda of >1 indicating long-term growth in this population (McElhany et al. 2007). 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 dam construction lacking passage facilities (McElhany 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 (McElhany et al. 2007).

Table 23. Upper Willamette river steelhead populations and a summary of available demographic data

Population[a]	Historical Abundance (Percent Annual change)[b]	Mean Number of Spawners (range)[c]	Long-term trend in redds per mile (95% CI)[d]	λ[e]
Mollala River	2,300 (-4.9)	914 (655-1275)	0.947 (0.918, 0.977)	0.988 (0.79, 1.235)
North Santiam River	2,000 (-4.0)	2,109 (1,485-2,994)	0.941 (0.906, 0.977)	0.983 (0.786, 1.231)
South Santiam River	550 (2.4)	2,149 (1,618-2,853)	0.936 (0.904, 0.907)	0.976 (0.855, 0.998)
Calapooia River	700	339 (206-560)	0.968 (0.933, 1.003)	1.023 (0.743, 1.409)

[a]Demographically independent populations identified by Myers et al. 2002 cited in Good et al. 2005.
[b]Values of historical abundance represent total escapement, with the exception of the Calapooia River which represents total run, as calculated in NMFS' first status review for the DPS. Data were collected using different methods (Angler Catch vs. Dam Counts) and represent data series ending in the early 1990s or earlier. Details on data types and the data series used for these calcuations are available in Busby et al. (1996).
[c]The geometric mean natural orgin spawner abudance calculated for the data series 1990-2005, and reported in McElhany et al. 2007.
[d]Long term trends are estimated using the entire data set, which is 1980 to 2000 for the Mollala River, and 1980-2001 for the remaining populations. Trends calculated by Good et al. 2005.
[e]Long-term growth rate (λ) reported by McElhany et al. 2007, and relects spawner escapement for the total available data series (1980-2005 – Molalla, Calappia & N Santiam Rivers; 1968-2005-S.Santiam River).

Critical Habitat

NMFS designated critical habitat for Upper Willamette River steelhead on September 2, 2005 (70 FR 52488). Designated critical habitat includes the following subbasins: Upper Willamette, North Santiam, South Santiam, Middle Willamette, Molalla/Pudding, Yamhill, Tualatin, and the Lower Willamette subbasins, and the lower Willamette/Columbia River corridor. These areas are important for the species' overall conservation by protecting quality growth, reproduction, and feeding. 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.

There are 38 watersheds within the range of Upper Willamette River steelhead. The total area of habitat designated as critical includes about 1,250 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 bankfull elevation. Of the 38 watersheds reviewed in NMFS' assessment of critical habitat for Upper Willamette River steelhead, 17 watersheds received a low rating of conservation value, six received a medium rating, and 15 received a high rating of conservation value for the species. In addition, the lower Willamette/Columbia River rearing/migration corridor downstream of the spawning range was rated as a high conservation value.

Marine Mammals

Cook Inlet Beluga Whale

Distribution and Description of the Listed 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). 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 in Hobbs et al. 2008; O'Corry-Crowe 2008; O'Corry-Crowe et al. 2008). Analysis of mtDN variation indicates strong philopatry to summering areas and low rates of dispersal between Cook Inlet beluga whales and other populations. The phylogenetic structure of the Cook Inlet beluga whale population suggests isolation of the population over evolutionary time scales.

Beluga whales are observed year-round in Cook Inlet although less is known about their winter movements than summer movements (see Hobbs et al. 2008 for a review). 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). 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 whale's preference for dense aggregations of preferred prey species.

Analyses of beluga whale stomach contents indicate that beluga whales are opportunistic feeders, 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 1989). 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. 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 1989). 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. 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).

Cook Inlet experiences some of the most extreme tidal fluctuations in the world (see NMFS 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. Aerial surveys and predictive models of habitat us indicate that beluga whale movement patterns are closely correlated to tidal patterns, flow accumulation and mudflats, with a preference for medium and high flow inlets of larger river basins (Ezer et al 2008; Goetz et al. 2007). More information, however, is needed to link these habitat attributes to causative reasons for this preference. Besides feeding, studies have suggested this preference for tidal mudflats may also be attributed to calving and breeding, molting, or shelter from predators like killer whales (Calkins 1989; Huntington 2000; Moore et al. 2000; Sheldon et al. 2003).

Beluga whale calving is not well documented but the presence of cow/calf pairs in large river estuaries in the upper inlet, and accounts of Alaskan Natives, suggests that calving and nursery areas are located near the mouths of the Beluga and Susitna Rivers, Chickaloon Bay and Turnagain Arm (NMFS 2008). According to surveys by LGL (Funk et al. 2005 as cited in NMFS 2008) 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). NMFS (2008) and others have suggested that the shallow waters of Cook Inlet may be important for reproduction and calving, as the shallower water is warmer which may confer an important thermal advantage for calf survival as they have relatively limited fat deposits at birth. Breeding is presumed to occur shortly after calving, in the late summer with a female's first parturition at age 5 or 6 after about 14-15 months of gestation (Calkins 1989). Lactation lasts about two years, with breeding occurring during lactation (Calkins 1989).

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 (See Hobbs et al. 2008 for discussion). Due to this ambiguity, Hobbs et al. (2008) summarized life history parameters according to tooth growth layers rather than years (Table 24 from Hobbs et al. 2008).

Table 24. 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 1986
	0% at 8-9 GLGs 33% at 10-11 GLGs 94% at 12-13 GLGs	Burns and Seaman 1986[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 1986[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 1986
Lifespan	>60 GLGs (Oldest female estimated at 70+ GLGs)	Burns and Seaman 1986
	64-65 GLGs	Khuzin 1961 (cited in Ohsumi 1979)
	60-61 GLGs	Brodie 1971
	50-51 GLGs	Sergeant 1973
Adult annual survival	0.96-0.97	Béland et al. 1992
	0.955 (based on pilot whale data)	Brodie et al. 1981
	0.935	Lesage and Kingsley 1998
	0.91-0.92	Allen and Smith 1978
	0.906 (includes natural & human-caused mortality)	Burns and Seaman 1986

Parameter	Data	Sources
	0.84-0.905 (based on body length and lifespan	Ohsumi 1979
Immature annual survival	0.905 (for neonates in first half year)	Sergeant 1973
Reproductive rate	0.010-012	Perrin 1982[c]
	0.11[d]	Burns and Seaman 1986
	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 1981
Calving Interval	<3 years	Burns and Seaman 1986[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."

Status

On October 22, 2008, NMFS listed the Cook Inlet beluga whale as endangered (73 FR 62919). Historic numbers of beluga whales in Cook Inlet are unknown. Dedicated surveys began in earnest in the 1990s when NMFS began conducting aerial surveys for beluga whales in Cook Inlet. Prior to then, survey efforts were inconsistent, part of larger sea bird and marine mammal surveys, made by vessel, or estimated following interviews with fishermen (Klinkhart 1966). In many cases the survey methodology or confidence intervals were not described. For instance, Klinkhart (1966) conducted aerial surveys in 1964 and 1965, where he describes having estimated the populations at 300-400 whales, but the methodology was not described nor did he report the variance around these estimates. Other estimates were incomplete due to the small area the survey focused upon (e.g. river mouth estimates; e.g., Hazard 1988). The most comprehensive survey effort prior to the 1990s occurred in 1979 and included transects from Anchorage to Homer, and covered the upper, middle and lower portions of Cook Inlet. From this effort, and using a correction factor of 2.7 to account for submerged whales Calkins (1989 cited in NMFS 2008) estimated the 1979 abundance at about 1,293 whales.

In 1993, NMFS began systematic aerial surveys of beluga whales in Cook Inlet and like the 1979 survey cover the upper, middle and lower portions of Cook Inlet. The survey protocol involves using paired observers who make independent counts at the same time a video of the whale grouping is recorded. Each group size estimate is corrected for subsurface and missed animals, or if video counts are not available then additional corrections are made (Allen and Angliss 2010).

Table 25. Estimated abundance of Cook Inlet beluga whales with coefficient of variation and 95% confidence intervals.

Year	Estimate[1]	CV	95% CI[2] Lower	95% CI[2] 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 1989 as cited in NMFS 2008.
[2]Data from R. Hobbs, pers. comm., to A. Garrett, Apr. 2010.

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). Given that harvest was curtailed significantly between 1999 and 2008, NMFS had expected the population would begin to recover at a rate of 2-6% per year. However, abundance estimates demonstrate that this is not the case (Hobbs & Shelden 2008).

In conducting its status review, NMFS ran a number of population viability analyses (PVAs) to estimate the time to extinction for Cook Inlet beluga whales. The models were sensitive to a variety of parameters such as killer whale predation, allee effects, and unusual mortality events. The best approximation of the current population incorporated killer whale predation at only one beluga whale per year, and allowed for an unusual mortality event occurring on average every 20 years. According to this model, 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).

Social Behavior

Beluga whales are highly social animals. The highly developed vocal repertoire of the beluga whale may play a substantial role in the formation of groups and communication among individuals. According to O'Corry-Crowe (2002), the beluga whale has long been called the "sea canary" by mariners because of the wide variety of sounds they make and can be heard reverberating through ship hulls. About 50 types of calls are recognized, typically ranging from

0.1 to 12 kHz, and include groans, whistles, buzzes, trills, roars and others, allow them to communicate over long distance and through icy arctic waters.

Belugas are typically observed in groups, which typically range from 2-25 individuals although they have been observed in groups of hundreds and even up to a thousand animals. There may be some seasonal segregation of sexes, as at times males form distinct groups and females are often tightly associated with one or more generations, at other times the groupings are a mixed social unit (O'Corry-Crowe 2002). Beluga whales also have a wide variety of facial expressions, as they can alter the shape of the mouth and melon. The lateral flexibility allows them to exploit shallow habitats and likely enhances visual signaling between animals (like vocalization, visual acuity is highly developed).

Threats

Natural Threats. Natural threats to Cook Inlet beluga whales include stranding, predation, parasitism and disease, environmental change, and genetic risks associated with small populations (e.g., inbreeding, loss of genetic variability). Beluga whales may strand accidentally as they occupy shallow water areas or escape predators, or as a result of diseases, illness or injury (NMFS 2008). Given the extreme tidal fluctuations in Cook Inlet, beluga whale strandings are not uncommon. According to NMFS (2008) killer whales have been observed in Cook Inlet concurrent to beluga whale strandings, and evidence of killer whale attacks is apparent in some beluga whale strandings (see Table 26).

According to NMFS (2008) over 700 beluga whales have stranded in Cook Inlet since 1988, many of which occurred in Turnagain Arm and often coincided with extreme tidal fluctuations (see Table 26 for a complete record). Where stranding occurs from extreme tidal fluctuations, and animals are out of the water for extended periods the risk of mortality increases from cardiovascular collapse. Ten hours may be the upper limit for out of the water for beluga whales before serious injury or death occurs (NMFS 2008). Strandings may represent a significant threat to the conservation and recovery of the Cook Inlet beluga whale population.

Table 26. Cook Inlet beluga whale stranding records from 1988 through September 2008 (from Hobbs and Shelden 2008, and NMFS 2008).

Year	Month	Location	No. w/evidence of Killer whale predation	Number of Whales	Known Associated Deaths	Total Mortalities* (live + dead stranded)
1988	October	Turnagain Arm		27	0	0
1989	-	-		-	-	4
1988	-	-		-	-	2
1991	August	Turnagain Arm	1	70-80	0	2
1992	October	Kenai River	2	2	2	5
1993	July	Turnagain Arm	1	10+	0	3
1994	June	Susitna River		186	0	7
1995	-	-		-	-	2
1996	June	Susitna River		63	0	12
	August	Turnagain Arm		60	4	
	September	Turnagain Arm		20-30	1	
	September	Knik Arm		1	0	

Year	Month	Location	No. w/evidence of Killer whale predation	Number of Whales	Known Associated Deaths	Total Mortalities* (live + dead stranded)
	October	Turnagain Arm		10-20	0	
1997	-	-		-	-	3
1998	May	Turnagain Arm		30	0	10
	September	Turnagain Arm		5	0	
1999	August	Turnagain Arm	5	58	5	12
	September	Turnagain Arm		12-13	0	
2000	August	Turnagain Arm	2	8	0	13
	September	Turnagain Arm		15-20	0	
	October	Turnagain Arm		1-2	0	
2001	-	-		-	-	10
2002	-	-		-	-	13
2003	April	Turnagain Arm	1	2	0	20
	August	Turnagain Arm		46+	5	
	September	Turnagain Arm		58	0	
	October	Turnagain Arm		9		
2004	-	-		-	-	13
2005	August	Knik Arm		6	1	6
2006	September	Knik Arm		12	0	8
2007	-	-		-	-	15
2008	August	Knik Arm	1	28-30	2	11

*Known subsistence harvested beluga whales are not included in these numbers.

Gaydos et al. (2004) identified 16 infectious agents in free-ranging and captive southern resident killer whales, but concluded that none of these pathogens were known to have high potential to cause epizootics. Many of these same infectious agents could pose a problem for Cook Inlet beluga whales. At this time little information is available to date to suggest bacterial or viral agents are actively contributing to the decline in the Cook Inlet population. About 80% of Cook Inlet beluga whales examined, however, have evidence of the parasite *Crassicauda giliakiana* in the kidneys, although it is presently unclear whether the parasite is affecting the status of the population (NMFS 2008). Necropsies have also revealed infestations of the common nematode anasakids, or whaleworm in the stomach of adult Cook Inlet beluga whales. While the parasite tends to favor the stomach and can cause gastritis or ulcerations, the infestations in beluga whales has not been considered severe enough to have caused clinical responses (NMFS 2008). Liver trematodes have also been identified in at least one beluga whale. At present, NMFS has no information to suggest that parasites are having a measureable impact on the survival and health of the Cook Inlet whale population (NMFS 2008).

Anthropogenic Threats. Human induced threats to Cook Inlet beluga whales include subsistence harvest, poaching and illegal harvest, incidental take during commercial fishing and reduction of prey through fishing harvests, pollution, oil and gas development, urban development, vessel traffic including from tourism and whale watching, noise, as well as research activities directed at beluga whales. Subsistence harvest of beluga whales by Alaskan natives has occurred since prehistoric times, but the effect of recent harvests has been significant. Although harvest levels have only recently been recorded, population declines in the mid 1990s are largely attributed to subsistence harvests during that period. In part, improved efficiencies of harvest techniques has allowed natives and others to increase catch of beluga whales. 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). Another commercial hunt of beluga whales in 1930s is recollected by residents, but no record of the hunt exists in Alaska fishery and fur seal documents (Bower, 1931-41 as cited in Mahoney and Shelden 2000). In 1999 and 2000 there was a voluntary moratorium on subsistence harvest, 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 (NMFS 2008).

Commercial fisheries likely have varying levels of interactions with Cook Inlet beluga whales, according to the timing, gear types, targeted species, and location of activities (NMFS 2008). Reports of fatal interactions with commercial fisheries have been noted in the literature (Murray and Fay 1979 cited in Hobbs et al. 2008; Burns and Seaman 1986). Direct interactions with fishing vessels and nets are considered unusual, based on observer data, and unlikely to inhibit the recovery of Cook Inlet beluga whales. The reduction of prey species, however, is of more concern for the species. In 2000 NMFS recommended the closing of the eulachon fishery due to a lack of understanding of how this fishery interfered with beluga whale feeding, but in 2005 this fishery was reopened with a harvest limited at 100 tons of eulachon. Currently, it is unclear if fishery harvest of beluga whale prey species is having a significant impact on the population. Impacts from recreational fisheries, which are very popular in the region, likely include the reduction of fish prey species particularly salmonid species, and also the harassment from noise and risk of injury from vessel strikes from the operation of small watercraft in the estuarine/river mouths may (NMFS 2008).

Contaminants in beluga whales are of concern, both for whale health and the health of subsistence users. Tissue samples are regularly collected from subsistence harvested and stranded beluga whales and archived. Tissues and organs commonly collected include blubber, liver and kidneys, as well as muscle, heart, bone, skin and brain. Blubber is the most commonly collected; due to the lipid content it typically contains the most lipophilic substances (Becker 2000). The kidney and liver are used to analyze heavy metal compounds. Relatively high levels of PCBs, chlorinated pesticides and mercury are evident in beluga whales, although the more contaminated belugas are from the St. Lawrence River, Canada (Becker 2000). Concentrations of chlorinated hydrocarbons in Cook Inlet beluga whales range from 0.1-2.4 µg/g, w.w. DDT, 0.6-4.7 µg/g, w.w. PCB, 0.1-0.6 µg/g, w.w. chlordane, <0.1-4.3 µg/g, w.w. toxaphene. The higher levels of these compounds found in beluga whales in comparison to bowhead whales is probably reflective of the trophic levels of the species, as bowhead are baleen whales that feed on copepods while belugas are primarily fish eaters (Becker 2000). Studies indicate that PCBs and chlorinated pesticide concentrations are higher in male beluga whales than females, reflecting the transference of body loads to the offspring that occurs during gestation and lactation (Becker et al. 2000). Other contaminant detected in Cook Inlet beluga whales include heavy metals such as cadmium, mercury, selenium, copper, and zinc to name a few. Comparative studies suggest that Cook Inlet beluga whales generally carry less body burdens than beluga whales from other areas. An exception is copper, which 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. 2000). To date, the health implications of high copper levels in Cook Inlet beluga whales is not clear.

Critical Habitat

NMFS proposed critical habitat for the Cook Inlet beluga whale on December 2, 2009 (74FR 63080). Two areas specific areas are proposed comprising 7,809 square kilometers of marine habitat. Area 1 encompasses 1,918 square kilometers (741 sq. mi.) of Cook Inlet northeast of a line from the mouth of Threemile Creek (61º 08.5'N., 151 º 04.4' W.) to Point Possession (61º 02.1'N., 150 º 24.3' W.). This area is bounded by Anchorage, the Matansuska-Susitna Borough, and the Kenai Peninsula Borough. This area contains shallow tidal flats, river mouths or estuarine areas and is important as foraging and calving habitats. Area 1 also has the highest concentrations of beluga whales in the spring through fall as well as the greatest potential for adverse impact from anthropogenic threats. Area 1 contains many rivers with large eulachon and salmon runs, including 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.

Southern Resident Killer Whale

Distribution and Description of the Listed Species

Three kinds of killer whales occur along the Pacific Coast of the United States: Eastern North Pacific (ENP) southern resident killer whales, ENP Offshore killer whales, and ENP transient killer whales. Of these only the southern resident killer whales are listed as endangered or threatened under the ESA. Southern resident killer whales primarily occur in the inland waters of Washington State and southern Vancouver Island, although individuals from this population have been observed off the Queen Charlotte Islands (north of their traditional range) and off coastal California in Monterey Bay, near the Farallon Islands, and off Point Reyes (NMFS 2005; BOR 2008).

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 regularly occur in the Georgia Strait, San Juan Islands, and Strait of Juan de Fuca (Heimlich-Boran 1988; Felleman et al. 1991; Olson 1998; Osborne 1999). The K and L pods typically arrive in May or June and remain in this core area until October or November, although both pods make frequent trips lasting a few days to the outer coasts of Washington and southern Vancouver Island (Ford et al. 2000). The J pod will occur intermittently in the Georgia Basin and Puget Sound during late fall, winter and early spring. 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).

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; Ford and Ellis 2006;). Throughout inland waters from May to September, southern resident killer whale diet is approximately 88% Chinook salmon, with a shift to chum salmon in fall. Chum salmon are also taken in significant amounts (11%), especially in autumn (Hanson et al. 2005; Ford and Ellis 2006; Hanson et al. 2007b). Chinook salmon 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 capture older (i.e., larger) than average Chinook salmon (Ford and Ellis 2006). 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 salmon, as well as salmon from Puget Sound, Washington and Oregon coasts, the Columbia River, and Central Valley of California (Hanson et al. 2007a). 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).

The local movements of southern resident killer whales usually follow the distribution of salmon (Heimlich-Boran 1986a, 1988, Nichol and Shackleton 1996). Areas that are major corridors for migrating salmon, and therefore, for southern resident killer whales, include Haro Strait and Boundary Passage, the southern tip of Vancouver Island, Swanson Channel off North Pender Island, and the mouth of the Fraser River delta, which is visited by all three pods in September and October (Felleman et al. 1991, Ford et al. 2000, K.C. Balcomb, unpublished data).

Female southern resident killer whales give birth to their first surviving calf between the ages of 12 and 16 years (mean ~ 14.9 years) and produce an average of 5.4 surviving calves during a reproductive life span lasting about 25 years (Matkin et al. 2003; Olesiuk et al. 1990). Females reach a peak of reproduction around ages 20-22 and decline in calf production gradually until reproductive senescence (Ward et al. 2009a). Older mothers tend to have greater calving success than do their younger, less-experienced counterparts (Ward et al. 2009b). Calving success also appears to be aided by the assistance of grandmothers (Ward et al. 2009b). The mean interval between viable calves is four years (Bain 1990). Males become sexually mature at body lengths ranging from 17 to 21 feet, which corresponds to between the ages of 10 to 17.5 years (mean ~ 15 years), and are presumed to remain sexually active throughout their adult lives (Christensen

1984; Duffield and Miller 1988; Olesiuk et al. 1990; Perrin and Reilly 1984). Most mating is believed to occur from May to October (Matkin et al. 1997; Nishiwaki 1972; Olesiuk et al. 1990). 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 200 kg (Clark et al. 2000; Ford 2002; Nishiwaki and Handa 1958; Olesiuk et al. 1990). Mothers and offspring maintain highly-stable, life-long social bonds and this natal relationship is the basis for a matrilineal social structure (Baird 2000; Bigg et al. 1990; Ford et al. 2000). Some females may reach 90 years of age (Olesiuk et al. 1990).

Southern resident killer whales spend a significant portion of the year in the inland waterways of the Strait of Georgia, Strait of Juan de Fuca, and Puget Sound, particularly during the spring, summer, and fall, when all three pods are regularly present in the Georgia Basin (defined as the Georgia Strait, San Juan Islands, and Strait of Juan de Fuca) (Felleman et al. 1991; Heimlich-Boran 1988; Olson 1998; Osborne 1999). Typically, K and L pods arrive in May or June and primarily occur in this core area until October or November. During this stay, both pods also make frequent trips lasting a few days to the outer coasts of Washington and southern Vancouver Island (Ford et al. 2000); however, J pod's movements differ considerably and are present only intermittently in the Georgia Basin and Puget Sound. Late spring and early fall movements of Southern Residents in the Georgia Basin have remained fairly consistent since the early 1970s, with strong site fidelity shown to the region as a whole (NMFS 2005b). During late fall, winter, and early spring, the ranges and movements of the southern residents are less well known. Offshore movements and distribution are largely unknown for the southern resident population.

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

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

Population Structure

Southern resident killer whale DPS consists of three pods, or stable familial groups: the J pod, K pod, and L pod. The J pod is seen most frequently along the western shore of San Juan Island and is the only pod observed regularly in Puget Sound throughout winter (Heimlich-Boran 1988; Osborne 1999). The K pod is most frequently observed during May and June when they occur along the western shore of San Juan Island while searching for salmon. The L pod is the largest of the three pods (Ford et al. 1994) and frequently breaks off into separate subgroups.

Status

Southern resident killer whales were listed as endangered under the ESA in 2005 (70 FR 69903). In the mid- to late-1800s, southern resident killer whales were estimated to have numbered around 200 individuals. By the mid-1960s, they had declined to about 100 individuals. As discussed in the preceding section, between 1967 and 1973, 43 to 47 killer whales were removed from the population to provide animals for displays in oceanaria and the population declined by about 30 percent as a result of those removals. By 1971, the population had declined to about 67 individuals. Since then, the population has fluctuated between highs of about 90 individuals and lows of about 75 individuals.

At population sizes between 75 and 90 individuals, we would expect southern resident killer whales to have higher probabilities of becoming extinct because of demographic stochasticity, demographic heterogeneity (Coulson et al. 2006; Fox et al. 2006) —including stochastic sex determination (Lande et al. 2003) — and the effects of phenomena interacting with environmental variability. Demographic stochasticity refers to the randomness in the birth or death of an individual in a population, which results in random variation on how many young that individuals produce during their lifetime and when they die. Demographic heterogeneity refers to variation in lifetime reproductive success of individuals in a population (generally, the number of reproductive adults an individual produces over their reproductive lifespan), such that the deaths of different individuals have different effects on the growth or decline of a population (Coulson et al. 2006). Stochastic sex determination refers to the randomness in the sex of offspring such that sexual ratios in population fluctuate over time (Melbourne and Hastings 2008). For example, the small number of adult male southern resident killer whales might represent a stable condition for this species or it might reflect the effects of stochastic sex determination. Regardless, a high mortality rates among adult males in a population with a smaller percentage of males would increase the imbalance of male-to-female gender ratios in this population and increase the importance of the few adult males that remain.

At these population sizes, population’s experience higher extinction probabilities because stochastic sexual determination leaves them with harmful imbalances between the number of male or female animals in the population (which occurred to the heath hen and dusky seaside sparrow just before they became extinct), or because the loss of individuals with high reproductive success has a disproportionate effect on the rate at which the population declines (Coulson et al. 2006). In general, an individual’s contribution to the growth (or decline) of the population it represents depends, in part, on the number of individuals in the population: the smaller the population, the more the performance of a single individual is likely to affect the population’s growth or decline (Coulson et al. 2006). Given the small size of the southern resident killer whale population, the performance (= “fitness,” measured as the longevity of individuals and their reproductive success over their lifespan) of individual whales would be expected to have appreciable consequences for the growth or decline of the southern resident killer whale population.

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

Social Behavior

Killer whales are highly social animals that occur primarily in groups or pods of up to 40-50 animals (Dahlheim and Heyning 1999; Baird 2000). Mean pod size varies among populations, but often ranges from 2 to 15 animals (Kasuya 1971; Condy et al. 1978; Mikhalev et al. 1981; Braham and Dahlheim 1982; Dahlheim et al. 1982; Baird and Dill 1996). Larger aggregations of up to several hundred individuals occasionally form, but are usually considered temporary groupings of smaller social units that probably congregate near seasonal concentrations of prey, for social interaction, or breeding (Dahlheim and Heyning 1999; Baird 2000; Ford et al. 2000).

In terms of gender and age composition, southern and northern resident killer whales social groups consisted of 19 percent adult males, 31 percent adult females, and 50 percent immature whales of either sex in 1987 (Olesiuk et al. 1990a). This composition is comparable with the composition of southern Alaska resident killer whales and killer whale populations in the Southern Ocean (Matkin et al. 2003; Miyazaki 1989).

Threats

Natural Threats. Southern resident killer whales like many wild animal populations (Nettles, 1992), experience highest mortality in the first year age class (Olesiuk et al. 1990; Krahn et al. 2002), although the reasons for these mortalities are still uncertain. The causes could include poor mothering, infectious or non-infectious diseases, and infanticide (Gaydos et al. 2004).

Gaydos et al. (2004) identified 16 infectious agents in free-ranging and captive southern resident killer whales, but concluded that none of these pathogens were known to have high potential to cause epizootics. They did, however, identify pathogens in sympatric odontocete species that could threaten the long-term viability of the small southern resident population.

Anthropogenic Threats. Several human activities appeared to contribute to the decline of southern resident killer whales. Southern resident killer whales were once shot deliberately in Washington and British Columbia (Scheffer and Slipp 1948; Pike and MacAskie 1969; Olesiuk et al. 1990; Baird 2001). Until 1970, about 25 percent of the killer whales that were captured for aquaria had bullet scars (Hoyt 1990). The effect of these attacks on individual whales or the population itself remains unknown. However, between 1967 and 1973, 43 to 47 killer whales were removed from the population for displays in oceanaria; because of those removals, the southern resident killer whale population declined by about 30%. By 1971, the population had declined to about 67 individuals. Since then, the population has fluctuated between highs of about 90 individuals and lows of about 75 individuals.

Over the same time interval, southern resident killer whales have been exposed to changes in the distribution and abundance of their prey base (primarily Pacific salmon) which has reduced their potential forage base, potential competition with salmon fisheries, which reduces their realized forage base, disturbance from vessels, and persistent toxic chemicals in their environment. The primary prey of killer whales, salmon, has been severely reduced due to habitat loss and overfishing of salmon along the West Coast (NRC 1996;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 kilometers of land. Contaminants entering Puget Sound and its surrounding waters accumulate in water, benthic sediments and organisms (Krahn et al. 2009). Exposure to contaminants may harm southern resident killer whales. The presence of high levels of persistent organic pollutants, such as PCB, DDT, and flame –retardants have been documented in southern resident killer whales (Ross et al. 2000; Ylitalo et al. 2001; Herman et al. 2005; Ross 2006). Although the consequences of these pollutants on the fitness of individual killer whales and the population itself remain unknown, in other species these pollutants have been reported to suppress immune responses (Kakushke and Prange 2007), impair reproduction, and exacerbate the energetic consequences of physiological stress responses when they interact with other compounds in an animal's tissues (Martineau 2007). Because of their long life span, position at the top of the food chain, and their blubber stores, killer whales would be capable of accumulating high concentrations of contaminants.

Since the 1970s commercial shipping, whale watching, ferry operations, and recreational boat traffic have increased in Puget Sound and the coastal islands of southern British Columbia. This traffic exposes southern resident killer whales to several threats that have consequences for the species' likelihood of avoiding extinction and recovering if it manages to avoid extinction. First, these vessels increase the risks of southern resident killer whales being struck, injured, or killed by ships. In 2005, a southern resident killer whale was injured in a collision with a commercial whale watch vessel although the whale subsequently recovered from those injuries. However, in 2006, an adult male southern resident killer whale, L98, was killed in a collision with a tug boat; given the gender imbalances in the southern resident killer whale population, we assume that the death of this adult male would have reduced the demographic health of this population (see further discussion below).

Second, the number and proximity of vessels, particularly whale-watch vessels in the areas occupied by southern resident killer whales, represents a source of chronic disturbance for this population. Numerous studies of interactions between surface vessels and marine mammals have demonstrated that free-ranging marine mammals engage in avoidance behavior when surface vessels move toward them. It is not clear whether these responses are caused by the physical presence of a surface vessel, the underwater noise generated by the vessel, or an interaction between the two (Goodwin and Green 2004; Lusseau 2006). However, several authors suggest that the noise generated during motion is probably an important factor (Blane and Jackson 1994; Evans et al. 1992, 1994). These studies suggest that the behavioral responses of marine mammals to surface vessels are similar to their behavioral responses to predators.

Several investigators have studied the effects of whale watch vessels on marine mammals (Watkins 1986; Cockeron 1995; Au and Green 2000; Erbe 2002; Félix 2001; Magalhães et al. 2002; Williams et al. 2002; Richter et al. 2003; Scheidat et al. 2004; Amaral and Carlson 2005; Simmonds 2005;). The whale's behavioral responses to whale watching vessels depended on the distance of the vessel from the whale, vessel speed, vessel direction, vessel noise, and the number of vessels. The whales' responses changed with these different variables and, in some circumstances, the whales did not respond to the vessels. In other circumstances, whales changed their vocalizations, surface time, swimming speed, swimming angle or direction, respiration rates, dive times, feeding behavior, and social interactions.

In addition to the disturbance associated with the presence of vessels, the vessel traffic affects the acoustic ecology of southern resident killer whales, which would affect their social ecology. Foote et al. (2004) compared recordings of southern resident killer whales that were made in the presence or absence of boat noise in Puget Sound during three time periods between 1977 and 2003. They concluded that the duration of primary calls in the presence of boats increased by about 15% during the last of the three time periods (2001 to 2003). At the same time, Holt et al. (2007) reported that southern resident killer whales in Haro Strait off the San Juan Islands in Puget Sound, Washington, increased the amplitude of their social calls in the face of increased sounds levels of background noise. Although the costs of these vocal adjustments remains unknown, Foote *et al*. (2004) suggested that the amount of boat noise may have reached a threshold above which the killer whales needs to increase the duration of their vocalization to avoid masking by the boat noise.

Critical Habitat

NMFS designated critical habitat for the DPS of Southern Resident killer whales 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 6,630 square kilometers of marine habitat. Three primary constituent elements 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, and 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.

Environmental Baseline

By regulation, the environmental baseline for biological opinions include the past and present impacts of all state, Federal or private actions and other human activities in the action area, the anticipated impacts of all proposed Federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of State or private actions which are contemporaneous with the consultation in process (50 CFR 402.02). The environmental baseline for this biological opinion also includes a general description of the natural factors influencing the current status of the listed species, their habitats, and the environment within the action area. The baseline analysis "is not the proportional share of responsibility the federal agency bears for the decline in the species, but what jeopardy might result from the agency's proposed actions in the present and future human and natural contexts." Pacific Coast Federation, 426 F.3d at 1093.

Our summary of the environmental baseline complements the information provided in the status of the species section of this Opinion, provides information on the past and present ecological conditions of the action area that is necessary to understand the species' current risk of extinction, and provides the background necessary to understand information presented in the *Effects of the Action* and *Cumulative Effects* sections of this biological opinion. The "impact" of the activities we normally identify in the *Environmental Baseline* of our Opinons allows us to assess the prior experience and state (or condition) of the endangered and threatened individuals and areas of designated critical habitat that occur in an action area. This is important because, as noted in the *Approach to the Assessment* section of this Opinion, in some phenotypic states, listed individuals will commonly exhibit responses they would not exhibit in other phenotypic states. The same is true for populations of endangered and threatened species: the consequences of change in the performance of individual on a population depend on the prior state of the population. Designated critical habitat is not different: under some ecological conditions, the physical and biotic features of critical habitat will exhibit response that they would not exhibit in other conditions. When we "add" the effects of a new, continuing, or proposed action to the prior condition of endangered and threatened individuals and designated critical habitat, as our regulations require, our assessments are more likely to detect a proposed action's "true" consequences on endangered species, threatened species, and designated critical habitat.

Because this is a programmatic consultation on what is essentially a continuing action with a geographic scope that encompasses all waters of the United States and its territories, this environmental baseline serves a slightly different purpose. First, as both a programmatic and a national consultation this Opinion does not assess the consequences of the EPA's recommended aquatic life criteria for specific sites or the listed resources that occur those specific sites. Rather, the *Environmental Baseline* for this Opinion focuses on the status and trend of the aquatic ecosystems in the United States and the consequences of that status for listed resources. 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 background information and context that is necessary for our assessment of the *Effects of the Action.*

We divided the environmental baseline for this consultation into five broad geographic regions of the United State: the Atlantic Northeast Region, the Atlantic Southeast 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 section we describe 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 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 resources.

Atlantic Northeast Region

This region encompasses Maine, New Hampshire, Massachusetts, Rhode Island, Vermont, Connecticut, New York, New Jersey, Delaware, Pennsylvania, Maryland, and Virginia. Major rivers in this region are the Penobscot, Connecticut, Hudson, Delaware, and Susquehanna rivers. Important estuarine areas include the Chesapeake Bay, Long Island Sound, Cape Cod Bay, and Massachusetts Bay.

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, and northern portions of the subtropical division – these ecoregions can be further subdivided into provinces based on vegetation. Climate is defined by hot humid summers and cold winters. Mean annual precipitation varies from about 35 to 45 inches per year. Vegetation in this region is characterized by tall broadleaf trees that provide a continuous dense canopy in summer, but shed their leaves completely in winter. Lower layers of small trees and shrubs are weakly developed. In spring, a luxuriant ground cover of herbs quickly develops, but is greatly reduced after trees reach full foliage and shade the ground. Needleleaf trees grow in colder, northern parts of the region and in mountain areas. Soils are generally rich in humus and strongly to moderately leached, although in the southern portions of this region, soils tend to be sandier and support second-growth forests of longleaf, loblolly, and slash pines (Bailey 1995).

Gulf of Maine

Natural History

This region encompasses drainages entering the Gulf of Maine, and is one of the most productive marine ecosystems in the world. Several significant rivers that drain into the gulf include the Merrimac, Kennebec, Androscoggin, Penobscot, and St. John Rivers (Table 24), and the significant estuaries that compose the larger Gulf of Maine include the Bay of Fundy, Massachusetts Bay, Merrymeeting Bay, and Cape Cod Bay. 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 strongly influenced by the Labrador Current, which makes the waters significantly colder and more nutrient rich than waters to the south, which are more strongly influenced by the Gulf Stream. The Gulf of Maine is characterized by salt marshes, kelp and seagrass beds, tidal mudflats, and underwater rocky outcrops, which form the foundation of a complex ecosystem and provide habitat for Atlantic herring, American lobster, Atlantic salmon, and several whale species. Merrymeeting Bay is the largest freshwater tidal estuary that enters the Gulf of Maine and has the largest freshwater outflow to the gulf (Kistner and Pettigrew 2001; Jackson et al. 2005). The

Kennebec and Androscoggin Rivers, along with four smaller tributaries, converge to form Merrymeeting Bay with the two larger rivers accounting for 98% of the inflow.

Table 27. Select rivers of the northeast United States that drain to the Gulf of Maine

Watershed	Approx. Length (mi)	Basin Size (mi^2)	Physiographic Provinces*	Mean Annual Precipitation (in)	Mean Discharge (cfs)	Number of Fish Species	Number of Endangered Species
Penobscot	275	8,592	NE	42	14,196	45	1 fish
Kennebec	230	5,383	NE	43	9,076	48	1 fish
Androscoggin	164	3,263	NE	44	6,180	33	1 fish
Merrimack	180	5,014	NE	36	8,299	50	1 fish

Data from Jackson et al 2005; Maine Rivers 2007a, b
*Physiographic Provinces: NE = New England, AD = Adirondack Mountains, VR = Valley Ridge, AP = Appalachian Plateau, PP = Piedmont Plateau, CP = Coastal Plain, BR = Blue Ridge.

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 (Table 25). While there is not much urban development in the Penobscot watershed except in and around Bangor, Doggett and Sowles (1989) report that 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 only major town in the Kennebec River watershed is Augusta, Maine (Jackson et al. 2005). The heaviest population density occurs in the watershed of the Merrimack River, which flows through industrial centers Manchester and Concord, New Hampshire, and Lowell and Lawrence, Massachusetts.

Textile mills, as well as paper and pulp mills, have long influenced water quality in the Penobscot, Kennebec, and Androscoggin rivers. The Kennebec 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, Harding Lawson Associates 2000). Since 1990, the levels of dioxins in other Maine rivers have been decreasing, but the levels in the Kennebec have remained constant (Kahl 2001). At one time, the Androscoggin River was considered one of the ten most polluted rivers in the country. The river has become much cleaner since the 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 one of the most heavily urbanized watersheds in the region, and some of the biggest sources of pollution facing the river are from industrial and urban sources, such as combined sewage overflows, industrial discharge, and stormwater run-off (USACE 2003). The upper mainstem of the Merrimack 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).

Toxins draining from river systems have produced significant toxin levels in regional estuarine systems, particularly from New Hampshire south throughout the Cape Cod region. Casco Bay still harbors residual sediment contamination and organic carbon levels from industries of a century ago, including heavy metals, PCBs, pesticides, TBT, dioxins and furans, and PAHs (EPA 2006). Low dissolved oxygen and red tide from nutrient loading also remain issues in the area. Habitats here remain relatively coalesced, although fragmentation is on the rise, and eelgrass beds have undergone local reductions.

Toxic sediments have been identified in Merrymeeting Bay, although some pollutants like metals declined in the bay between 1980 and 1991, although copper levels have increased (Hayden 1998). Sediments associated with the Androscoggin River exhibit higher levels of PAHs and mercury, while sediments from the Kennebec River had higher levels of chromium, arsenic, and selenium (Hayden 1998). Merrymeeting 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. 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).

Human activities have impact the coasts of New Hampshire and Massachusetts. New Hampshire estuaries suffer from habitat fragmentation and degradation, bacterial and nutrient contamination, salt marsh degradation, and declines in the commercially valuable oyster and clam populations resulting from sewage and industrial pollution (EPA 2006). Several areas experience elevated nitrogen and phosphorus in water, high total organic carbon, and sediment contaminant levels in the benthos, as well as above average contaminants (PAHs, DDT, and PCBs) in fish and shellfish. A massive decline in eelgrass habitats occurred in 1989 and meadows have been relatively constant since.

Estuarine and bay systems of Massachusetts experience pressures from the major metropolitan region around Boston Harbor. The increased sewage and stormwater outflow results in a loss of roughly 1,000 acres of wetland habitat per year and cause closings in shellfish harvests due to bacterial contamination. Local wetland restoration projects have improved over 450 acres of wetland in the region. Over 26 invasive species have been identified in Massachusetts Bay, including the Asian shore crab and Pacific tunicate, and have contributed to a reduction in the industrial scallop fishery.

Table 28. Land uses and population density of several watersheds that drain to the Gulf of Maine

Watershed	Land Use Categories (%)				Density
	Agriculture	Forested	Urban	Other	(people/mi^2)
Penobscot	5	95			21
Kennebec	6	82	2	10	39
Androscoggin	5	86	2	7	65
Merrimack	6	75	13	6	404

Data from Jackson et al. 2005

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. Because 6,000 to 10,000 salmon are required for a sustainable population, the Penobscot run depends on fish from a local hatchery (Moore and Platt 1996).

The Kennebec River mainstem has eight large hydroelectric dams, which restrict 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, dissolved oxygen levels and macroinvertebrate density have improved. Additionally, in 2007, the fish passage facilities on the lowest dam on the Kennebec River, as well as those on the Sebasticook River’s second and third lowest dams, became operational.

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 dissolved oxygen throughout the summer. During the 1960s, most of the river had oxygen levels of 0 ppm, 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 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 watersheds of the Atlantic Northeast Region began before 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 watersheds that have become Superfund sites due to excessive pollutants being leached into groundwater, such as Elizabeth, Pike Hill, and 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). Many of the abandoned mines are scheduled for cleanup; however, the impacts of their former use could persist for years after decommissioning.

*Commercial and Recreational Fishing**.*** The primary commercial fisheries along the Northeast 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).

Long Island and the Connecticut River

Natural History

South of the Gulf of Maine is the Long Island Sound watershed, which 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 fresh water from the Connecticut, Thames, and Housatonic rivers (90% of the freshwater input) mixes with the Atlantic Ocean. The sound ranges in salinity from 23 ppt in the western end to 35 ppt on the eastern side. The surface area of Long Island Sound is 1,320 square miles, draining an area of over 16,000 square miles. 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 Long Island Sound as spawning grounds.

The Connecticut River drains a watershed of 11,259 square miles 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 bed is composed of glacial deposits and granitic bedrock. The average annual precipitation is approximately 43 inches. At the mouth, the average discharge is 10.2 billion gallons per 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 fresh water 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). Much of the lead, copper, and zinc are likely 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 dissolved oxygen 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 dissolved oxygen levels above 5 ppm for surface water, above 3.5 ppm for deeper water, and at or above 2 ppm for all water.

Within the Connecticut River watershed the dominant land use is forest (80%), with 11% used for agriculture and the remaining 9% in mixed uses (Jackson et al. 2005). Major towns in the Connecticut River 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).

Estuaries within Long Island Sound have historically been plagued by low dissolved oxygen, pathogens, habitat degradation and species decline, and sediment contamination (EPA 2006). These issues remain relevant today, with increasing human populations increasing contaminant loads and decreasing wetland habitat. Almost all measures of quality have been affected, including phosphorus load, low dissolved oxygen, and chlorophyll *a* concentrations, high sediment contaminants (DDT and metals) and total organic carbon, as well as excessive levels of PCBs in nearly all fishes sampled. Riverine and wetland restoration has been ongoing for several years and provided an additional 2,000 acres of wetland and over 50 miles of stream passage for migratory fishes. This may help curtail the decline of estuarine bird populations and oysters in recent years. Oyster harvest closures resulting from pathogen concentrations have been common for two decades and additional regulation of vessel discharges, illegal sewage connections to Long Island Sound, high volume of storm water effluent, and malfunctioning septic systems are identified as point sources for this.

Hydromodification Projects. The Connecticut River has 16 hydroelectric dams on the mainstem of the river and as many as an estimated 900 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 dissolved oxygen 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. Towns such as Plymouth, Vermont were famous for mining gold, iron, talc, soapstone, marble, asbestos, and granite (Ewald 2003). Other towns throughout New Hampshire and Vermont also mined gold, silver, soapstone, talc, granite, slate, and copper (Ewald 2003). There are many mines along the

Connecticut River, which currently degrade the river's water quality, including the country's first chartered copper mine. 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 although there has been and continues to be discussions about mining for sand and gravel.

Commercial and Recreational Fishing. Few commercial fisheries exist in the Connecticut River. Shad is the primary commercial fishery, 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 others.

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 dissolved oxygen deficiencies, the western portion of Long Island Sound has seen major declines in fish and shellfish populations. Despite these declines, the sound houses the largest oyster fishery in the US, providing 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. Weber (1996) reported that these incidental captures caused abandonment of spawning migrations in the Ogeechee River, Georgia.

Hudson River

Natural History

The Hudson River flows approximately 315 miles to the ocean, with a watershed of 13,365 square miles. The river flows from the Adirondack Mountains, draining most of eastern New York State, to the Atlantic 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 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. The lower portion of the watershed is a mixture of sedimentary, metamorphic, and igneous rocks. Average annual precipitation is approximately 36 inches per year. At the mouth, the average discharge is 13.5 billion gallons per day, or 20,906 cubic feet per second (Jackson et al. 2005). The Hudson River is a freshwater tidal estuary between Troy, New York 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 as well as Jersey City, New Jersey. The average population density in the watershed is approximately 350 people per square mile, but the actual density within a reach of the watershed varies widely. For instance, according to Jackson et al. (2005) population density at the headwaters at Lake Tear of the Clouds is 0 people per square mile, while the population density in Manhattan is over 25,907 people per square mile.

Throughout the 20th century, power plants, municipalities, pulp and paper mills, and corporations such as IBM, General Motors, and General Electric in particular, whom 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 River. The PCB levels in the Hudson River are among the highest nationwide. The upper basin is mostly unaffected by humans, with clear, soft water with low nutrients. The middle Hudson River 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 river 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 River 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, zinc, DDT, PCBs, and PAHs in quantities above the EPA recommended levels in sediments and fish tissue throughout the watershed (Wall et al. 1998).

Estuarine conditions surrounding the Hudson River are extremely poor (EPA 2006). Most issues stem from the extremely dense human population center around New York City. Fish consumption warnings are commonplace due to high mercury levels, over 200 million gallons of untreated sewage enter the bay daily, and only 20% of the former wetland area remains. Nitrogen and phosphorus are generally very high, sediments are highly contaminated (PCB), and total organic carbon is generally elevated. Only about 20,000 acres of wetland remain in the region. Although these poor conditions persist, wading birds formerly absent are present today and osprey (indicator bird species) are showing resurgence.

Hydromodification Projects. The mainstem Hudson River has 14 dams and dams exist 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 has a low number of physical obstructions, and contains one of the largest populations of endangered shortnose sturgeon in the United States. Prior to the Clean Water Act, the middle stretch of the Hudson River 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 (Jackson et al. 2005).

Mining. The Hudson River has been periodically important as a source of metals and mined resources. The Adirondack Mountains, in the headwaters, have been mined for 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 and mined until 1982, when it was abandoned.

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, New York 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 River.

Delaware River

Natural History

The Delaware River flows approximately 329 miles to the ocean, with a watershed of 12,757 square miles. 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 annual precipitation is approximately 43 inches. At the mouth, the average discharge is 9.6 billion gallons per day, or 14,903 cubic feet per second. Although it is only the 42nd largest river in the United States 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 eight 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 with most distributed around the estuary (Fischer et al. 2004; Jackson et al. 2005). As the area's population grew, water quality significantly degraded around Philadelphia with serious water quality problems observed as early as 1799. By the 1960s, the average dissolved oxygen in the lower river was approximately 0.2 ppm. A survey in the 1970s of organochlorine frequency in rivers ranked the Delaware at Trenton and the Schuylkill, the largest tributary to the Delaware River, as the 8th and 1st worst, respectively in the nation (Jackson et al. 2005). Urban and agricultural activities continue to affect the basin water quality today. Organochlorines, pesticides, nutrients, organics, and trace elements were widely detected in small tributaries and mainstream reaches (Fischer et al. 2004). In the Delaware River Basin, commonly detected organochlorines include DDTs, PCBs, chlordanes, and dieldrin. Fisher et al. (2004) found that 84% of the fish tissue sampled contained PCBs, while 21% of the sediment samples contained PCBs despite that the manufacture and use of these compounds ceased in the 1970s or earlier. These compounds are bioaccumulating in the food chain, and occasionally exceed wildlife protection guidelines (52% of the sites exceeded wildlife guidelines for PCBs, whereas 16% of the sites exceeded guidelines for dieldrin, and 12% exceeded wildlife guidelines for DDT [Fischer et al. 2004]).

Trace metal contamination is also a significant concern within the basin, and may be a particular concern for benthos including endangered shortnose sturgeon. Trace metals detected at high levels included arsenic, cadmium, chromium, copper, lead, mercury, nickel, and zinc (Fischer et al. 2004). Most trace metal contamination was associated with former or on-going industrial sites such as mines and metal smelters.

The Delaware Estuary is considered to be in poor condition primarily from historical and modern toxin contributions from population centers such as Philadelphia along the Delaware River (EPA 2006). Overfishing and habitat loss also play a role in and along the estuary system itself. Estuarine waters contain high nitrogen and phosphorus levels with low chlorophyll *a* concentrations. Water clarity is variable, but fish tissues contain unsatisfactory levels of PCBs, DDT, and PAHs. Invasive plant species, including the common reed and purple loosestrife, have displaced native marsh species. Disease and low recruitment in oyster populations have had significant effects in commercial and ecological parameters of Delaware Bay. Shad populations declined due to poor water quality and have not recovered, which may indicate environmental stress in several other finfish populations.

Hydromodification Projects. The Delaware River has 16 dams in its headwaters, but the middle and lower rivers are 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.

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.

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.

Chesapeake Bay lies totally within the Atlantic Coastal Plain but the watershed includes parts of the Piedmont Province and the Appalachian Province. 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. More than 100 rivers and streams deliver fresh water to Chesapeake Bay, and major rivers include the Susquehanna and Potomac (see Table 26).

The Susquehanna River, rated as the 18th largest river in the United States based on average discharge at the mouth, flows approximately 448 miles to Chesapeake Bay (Kammerer 1990; Jackson et al. 2005). The river flows north to south from New York, through Pennsylvania, and reaches the Chesapeake Bay at 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 Susquehanna drains into the most northern portion of Chesapeake Bay and is not tidally influenced. The Susquehanna River provides about 50% of the freshwater inflow into the Chesapeake Bay.

Table 29. Select rivers of the northeast United States that drain to Chesapeake Bay

Watershed	Approx. Length (mi)	Basin Size (mi^2)	Physiographic Provinces*	Mean Annual Precipitation (in)	Mean Discharge (cfs)	No. Fish Species	No. Endangered Species
Susquehanna	448	27,580	AP, VR, PP, BR	39	40,718	103	2 birds
Potomac	383	14,700	AP, VR, PP BR, CP	39	11,301	95	1 fish, 1 mussel
James	340	10,102	VR, BR, PP, CP	43	8,016	109	3 fish, 1 amphibian, 1 reptile, 6 mussels

Data from Jackson et al. 2005; Smock et al. 2005
*Physiographic Provinces: NE = New England, AD = Adirondack Mountains, VR = Valley Ridge, AP = Appalachian Plateau, PP = Piedmont Plateau, CP = Coastal Plain, BR = Blue Ridge.

The Potomac and James Rivers, on the other hand, are located south of the Susquehanna River basin and are tidally influenced. The Potomac River estuary begins two miles below the Washington, D.C.-Maryland border, just below the Little Falls of the Potomac River. The river's headwaters begin in the Allegheny Mountains of West Virginia and the Potomac and 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. Ninety-five fish species live in the Potomac, but only 65 of those are native to the area (Jackson et al. 2005). The James River is one of the longest bodies of water in entirely one state, beginning in the Allegheny Mountains of western Virginia and flowing across the state to the Chesapeake Bay.

Human Activities and Their Impacts

Land Use. The Chesapeake Bay area, while dominated by forested lands, is more heavily influenced by agriculture than many other areas in the northeast and middle Atlantic (see Table 27 for land uses by watershed). Urbanized areas are scattered but dominate the landscape in certain areas (e.g., Washington D.C. metro area, in the Potomac River watershed, and Scranton and Harrisburg, Pennsylvania in the Susquehanna River watershed). Most of the bay's waters are degraded, and algal blooms are a chronic problem. Nutrient pollution and heavy sediment loads have lead to large anoxic areas within the bay, and fish kills in some areas. Agricultural practices, urban development, as well as natural sources of sediment influence water quality within the bay. Past logging practices in the basins draining to Chesapeake Bay also influenced sediment loads within several rivers. In the Susquehanna River watershed, sediment transport in the early 1900s was nine times higher than it was 200 years earlier, due to logging and agriculture.

Overall, in 2006, less than one-third of Bay water quality goals were met (Chesapeake Bay Program 2007). Direct discharges of sewage and industrial wastewater into the Susquehanna River watershed and contributes to degraded water quality in the basin. The number of outfalls totals over 400 in the watershed, generally with the number of outfalls being proportional to the size of the city (Binghamton, New York: 10, Harrisburg, Pennsylvania: 65, Scranton, Pennsylvania: 70). As a result, the Susquehanna River 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 River was named America's most endangered river by American Rivers, who produces an annual list. Even 35 years after the Clean Water Act, there are still elevated levels of copper, sulfur, selenium, arsenic, cobalt, chromium, lead, mercury, zinc, and pesticides (Beyer and Day 2004) as well as depressed pH associated with abandoned mines in the watershed (Hoffman 2008). Excessive nutrient and sediment loads also significantly impair the Susquehanna, stemming from urban and agricultural runoff and sewage treatment discharge (Hoffman 2008). Although water quality has significantly improved in the Potomac River over the past 50 years, the river remains threatened by 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.

Similarly, the James River has elevated levels of zinc, copper, cadmium, nickel, chromium, lead, arsenic, dioxin, PCBs, and pesticides. The James River was also the site of one of the nation's most publicized pollution events when a manufacturing plant discharged, for nearly ten years, the chlorinated insecticide Kepone in the lower river (Smock et al. 2005). The insecticide is bioaccumulated, and resulted in a ban on commercial fishing in the lower river. Although the ban has been lifted, accumulations of Kepone in the riverine sediments are still cause for concern (Smock et al. 2005).

Table 30. Land uses and population density in several watersheds that drain to Chesapeake Bay

Watershed	Land Use Categories (%)				Density (people/mi^2)
	Agriculture	Forested	Urban	Other	
Susquehanna	27	63	9	---	145
Potomac	32	58	5	6	358
James	23	71	6	---	249

Data from Jackson et al. 2005; Smock et al. 2005

Hydromodification Projects. The Chesapeake Bay is home to several moderate to small sized dam projects. While only a few impoundments in the Potomac River and its tributaries are larger than 1.5 square miles, the Susquehanna River has over 100 dams along the mainstem and the first major dam is located just 10 miles upstream of the mouth. The Anacostia River, a major tributary to the Potomac River is scheduled to have some 60 dams removed or altered to improve water quality and fish passage. Dams in other basins have also been upgraded or are planned for upgrades. For instance, between 1928 and 1972, no shad passed Conowingo Dam on the Susquehanna River, 10 miles upstream of the mouth, but since passage facilities were installed fish abundance has increased from approximately 100 to more than 100,000 individuals. Similarly, many dams have been improved or removed in the James River. In 1999, a fish ladder added to Boscher Dam, which had prevented upstream fish runs since 1823 provided access to 137 additional miles of the James River and 168 miles of its tributaries.

Mining. In the Chesapeake Bay watershed, coal mining has likely had the most significant impact on water quality. 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 was harvested from behind one dam. Later, between 1951 and 1973, over 10 million tons were harvested from behind another dam. 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 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. 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. Many of these abandoned coal mines must be treated with doses of limestone to balance the pH of the water draining from the mines. Coal is abundant through the watershed, amounting to nearly 30 billion tons of coal mined. Coal is no longer a primary industry in the watershed, but the impacts of the acid mine drainage are still prominent.

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). Populations 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).

Atlantic Southeast Region

This region covers all drainages that ultimately drain to the Atlantic Ocean (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 its winter deciduous forests 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 Atlantic Ocean (Bailey 1995).

Most of the Atlantic Southeast 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 Ocean. 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 Rivers. The more northern 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 et al. 2005). We describe 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 their 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 2006). The Estuarine Complex is separated from the Atlantic Ocean by the Outer Banks, a long barrier peninsula, and is characterized by shallow waters and 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. Basin geology 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. 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 et al. 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 more than half of the water flowing in streams discharging to the Albemarle-Pamlico 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 square miles. The Roanoke River begins in the mountains of western Virginia and flows across the North Carolina border before entering 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 river is almost entirely organic-rich mud. The average annual precipitation is approximately 43 inches. At the mouth, the average discharge is 5.3 billion gallons per day, or 8,193 cubic feet per second (Smock et al. 2005). The Roanoke River is home to 119 fish species, and only seven of those are not native to the area (Smock et al. 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 square miles (Smock et al. 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 annual precipitation is approximately 48 inches. At the mouth, the average discharge is 3.4 billion gallons per 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.

Three percent of the basin qualifies as urban land uses and 25% is used for agriculture (Smock et al. 2005). The only major town in the Roanoke watershed is Roanoke, Virginia and population in the watershed is approximately 80 people per square mile (Smock et al. 2005). In contrast, the Neuse River watershed is described as 35% agriculture, 34% forested, 20% wetlands, 5% urban, and 6% other, with a basin-wide density of approximately 186 people per square mile (Smock et al. 2005). While the population has 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 2006). Much of the estuarine complex is protected by large tracts of state and federally protected lands, which may reduce development pressures.

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 contribute to 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 as cited in EPA 2006) 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.

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 freshwater 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 dissolved oxygen concentrations and sudden wind changes that mix hypoxic waters, are some of contributing factors or causes to the basins persistent fish kills (NCDWQ 2004).

Both the Roanoke River and the Neuse River 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 nationally (Spruill et al. 1998).

Conditions within the Albemarle-Pamlico estuary system are relative good compared to other northeastern estuaries. Agricultural and urban runoff provide the majority of toxins to the region that can impair water and habitat quality, as well as degrade fishery resource quality and quantity, including Atlantic sturgeon and numerous sport and commercially important fish species (EPA 2006). Chlorophyll *a* is the most significant detractor to water quality and total organic carbon has the greatest impact on sediment quality. Benthic quality and fish tissue contamination (PAHs and PCBs) also have suffered from human-introduced toxins. Losses of 25 to 50% of wetlands surrounding tributaries have lead to significant reduction in habitat as a result of human development.

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 these methods can 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 28 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 in Georgia and the St. Johns 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) described the mountains and plateau as heavily dissected and 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. At the fall line, where the Piedmont Plateau meets the sedimentary deposits of the Coastal Plain, steep changes in elevation result in rapids or falls before the rivers level off in their Coastal Plain reaches. Here, 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 rivers that originate in the Piemont Plateau) appearance. The Satilla River is a blackwater river that has a naturally low pH (between four and six) 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 31. Rivers of the Southeast United States

Watershed	Approx Length (mi)	Basin Size (mi^2)	Physiographic Provinces*	Mean Annual Precip (in)	Mean Discharge (cfs)	No. Fish Species	Number of Endangered Species
Cape Fear	320	9,324	PP, CP	47	7,663	95	8 fish, 1 mammal, 15 mussels
Great Pee Dee	430	10,641	BR, PP, CP	44	13,102	>100	6 fish, 1 reptile
Santee-Cooper	440	15,251	BR, PP, CP	50	15,327	>100	5 fish, 2 reptiles
Savannah	300	10,585	BR, PP, CP	45	11,265	>100	7 fish, 4 amphibians, 2 reptiles, 8 mussels, 3 crayfish
Ogeechee	250	5,212	PP, CP	44	4,061	>80	6 fish, 2 amphibians, 2 reptiles, 1 mussel
Altamaha	140	14,517	PP, CP	51	13,879	93	1 mammal, 12 fish, 2 amphibians, 2 reptiles, 7 mussels, 1 crayfish
Satilla	200	3,530	CP	50	2,295	52	2 fish, 1 amphibian, 2 reptiles, 1 mussel
St. Johns	311	8,702	CP	52	7,840	>150	1 mammal, 4 fish, 2 reptiles, 2 birds

Data from NCDENR 1999; Smock et al. 2005
*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 et al. 2005). See Table 29 for a summary of land uses and population densities in several area basins across the region (data from Smock et al. 2005).

The largest population centers in the region include Miami and Jacksonville, Florida and Savannah, Georgia. Major towns include Greensboro, Chapel Hill, Fayetteville, 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, ammonia, turbidity, and low dissolved oxygen. These impairments are caused by municipal sewage overflows, mining, non-point source pollution, waterfowl, urban runoff, marinas, agriculture, and industries including textile manufacturing, power plant operations, paper mills, and chemical plants (Mehta 2008; Harned and Meyer 1983; Berndt et al. 1998; NCDENR 1998; Smock et al. 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 stream flow, which could be related to downstream transport by subsurface flows (Berndt et al. 1998).

Table 32. Land uses and population density in several Atlantic southeast basins

Watershed	**Land Use Categories (%)**				**Density**
	Agriculture	**Forested**	**Urban**	**Other**	**(people/mi^2)**
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

Data from Smock et al. 2005

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%. The reduction in sediment inputs from farms did not translate to a reduction in sediment in the river, and during this period there was a 25% reduction in agricultural land and a 38% increase in urban development.

Where data are available, estuaries throughout the region contain generally moderate to severe nitrogen loads from river systems (Bricker et al. 2007). This has lead to toxic blooms of algae in some areas. Eutrophication has been noted particularly in the St. Johns River region. Low dissolved oxygen levels have also been found in the area around the Savannah River.

Mining. Mining occurs throughout the region. South Carolina is ranked 25th 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 South Carolina's leading nonfuel minerals in 2004 (NMA 2007). In contrast, Georgia accounts for 4%, Florida accounts for 5%, and North Carolina accounts for about 2% of the total non-fuel 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 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. Mines release toxic materials that negatively affect fish, as fish living around dredge tailings have elevated levels of mercury and selenium.

Hydromodification Projects. Several area rivers have been modified by dams and impoundments. In contrast to rivers along the Pacific Coast, considerable less information is available 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 numerous 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 Atlantic 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 reservoir. Although the Altamaha River is undammed, hydropower dams are located on its tributaries, the Oconee and Ocmulgee rivers, above the fall lines. There are no dams 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 by water diversions for drinking water and agriculture.

Commercial and Recreational Fishing. The region is home to many commercial fisheries targeting 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. There are approximately 1.15 million recreational anglers in the state as well.

Southwest Coast Region

The basins described in this section 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 30).

Table 33. Select rivers in the southwest coast region

Watershed	Approx Length (mi)	Basin Size (mi^2)	Physiographic Provinces*	Mean Annual Precipitation (in)	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	3,651	PB	52	7,416	25 (15)	12
Russian River	110	1,439	PB	41	2,331	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	4,241	PB	14	448	36 (16)	42 T & E spp.
Santa Ana River	110	2,438	PB	13	60	45 (9)	54
Santa Margarita River	27	1,896	LC, PB	49.5	42	17 (6)	52

Data from Carter and Resh 2005
*Physiographic Provinces: PB = Pacific Border, CS = Cascades-Sierra Nevada Range, B/R = Basin & Range.

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 caused 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 Range.

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 topography 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 unpredictable within a basin, but consistant 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.

The San Joaquin River, draining the largest basin in the region, originates within the Sierra

Nevada Range near central 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 River 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 31). 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. 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 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 1998; Belitz et al. 2004).

Table 34. Land uses and population density in several basins of the southwest coast region

Watershed	**Land Use Categories (%)**				**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

Data from Carter and Resh 2005

Not only is the Santa Ana watershed the most heavily developed watershed 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 (USGS) 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 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 (Leenheer et al. 2008; Kent et al. 2005; 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 chloroform, which sometimes exceeded water quality standards), pesticides (diuron, diazinon, carbaryl, chlophyrifos, lindane, malathion, and chlorothalonil), and trace elements (lead, zinc, and arsenic). As a result of the changes, the biological community in the basin is heavily altered (Belitz et al. 2004).

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 over 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 influenced by intensive agriculture; water hardness, alkalinity, nutrients and conductivity are high in areas where agricultural uses predominate.

Estuary systems of the region are consistently exposed to anthropogenic pressures stemming from high human density sources. As an example, the largest west coast estuary, the San Francisco Estuary, provides drinking water to 23 million people, irrigates 4.5 million acres of farmland, and drains roughly 40% of California’s land area. As a result of high use, many environmental measures of the estuary are poor. Water quality suffers from high phosphorus and nitrogen loads, primarily from agricultural, sewage, and storm water runoff. Water clarity is also compromised. Sediments contain high levels of the contaminants PCB, pesticides, mercury, copper, and silver from urban runoff and historical activities. As these persist in the marine environment, the estuary system will likely carry loads for years to come, even with strict regulation or banning. Gold mining has reduced estuary depths in much of the region, causing drastic changes to habitat. Large urban centers are foci for contaminants and levels near San Francisco, Oakland, and San Jose are highest and are also where water clarity tends to be at its worst. These water and sediment quality characteristics biomagnify into the food chain; fish tissues contain high levels of particularly PCB and mercury, the former being concentrated 10 times more than human health guidelines for consumption. Birds, some of whom are endangered (clapper rail and least tern), have also concentrated these toxins.

Invasive species have become an increasingly recognized issue. Giant reeds have displaced native marsh species in many areas. Marine invasive species include the green crab, shimofuri goby, Asiatic clams, and zooplankton; these species are cited in reducing the abundance of local species. The Asian clam has become the dominant infaunal species and has likely reduced primary production in the estuary system (Nichols et al. 1990; Ray 2005).

Red tide significantly affects the California coastline. Here, poisoning and mortality of California sea lions, fish, and birds have been recorded, the most recent of which was in 2007 (Chea 2007). California red tide events are correlated with El Niño oscillations. In addition to the toxin produced by red tide diatoms, a pathogen associated with cholera has been identified in California red tide blooms (Mouriño-Pérez et al. 2003).

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 in 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 *in* Carter and Resh 2005). While about 75% of the runoff occurs in basins in the northern half of California, 80% of the water demand is in the southern half. 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 per 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 roughly 22 million residents and growing.

Both the Sacramento and San Joaquin rivers are heavily modified, each with hundreds of dams. In 2009, the Sacramento-San Joaquin river system was named America's most endangered river by American Rivers. In the prior year, the Rogue River was listed as the second most endangered river. 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 each. The Santa Margarita, considered one the last free flowing rivers in coastal southern California has nine 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, and the temperature regime and sediment transport regime are altered. Such changes have had profound influences on the biological community within the basin. These modifications generally result in a reduction of suitable habitat for native species and frequent concomitant increases in 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).

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 total production (NMA 2007). Today, gold, 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).

California contains some 1,500 abandoned mines and roughly 1% are suspected of discharging metal-rich waters in the basins. The Iron Metal Mine in the Sacramento Basin releases more than 1,100 pounds of copper and more than 770 pounds of zinc to the Keswick Reservoir below Shasta Dam, as well as elevated levels of lead (Cain et al. 2000 *in* Carter and Resh 2005). Metal contamination seriously reduces the biological productivity within a basin and can result in fish kills at high levels or sublethal effects at low levels, including reduced feeding, overall activity levels, and growth. The Sacramento Basin and the San Francisco Bay watershed is one of the most heavily affected 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 five times lower than concentrations detected within San Francisco Bay today (Conaway et al. 2003 in EPA 2006).

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. 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 (NRC 1996).

Pacific Northwest Region

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 Division, Marine Division – Mountain Provinces, Temperate Steppe Division, Temperate Steppe Division – Mountain Provinces, and portions of the Temperate Desert Davison. 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 basin within the region is the Columbia River. The largest river in the Pacific Northwest and the fourth largest river in terms of average discharge in the United States, it drains 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 et al. 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 32 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) and the 19th largest river in the nation in terms of average annual discharge (Kammerer 1990). The basins drain portions of the Rocky Mountains, Bitteroot Range, and the Cascade Range.

The average annual discharge at the mouth of the Columbia River is 265,000 cubic feet per second (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 35. Select tributaries of the Columbia River

Watershed	Approx Length (mi)	Basin Size (mi^2)	Physiographic Provinces*	Mean Annual Precip. (in)	Mean Discharge (cfs)	No. Fish Species (native)	No. Endangered Species
Snake/Salmon	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	214	6,139	CS, CU	7	3,602	50	2 (T)
Willamette	143	11,478	CS, PB	60	32,384	61 (~31)	5 fish (4 T, 1 E),

Data from Carter and Resh 2005
*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 with unique adaptations to local environments within a tributary (Stanford et al. 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 1979). According to a review by Stanford et al. (2005), the basin formerly 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 1979).

Human Activities and Their Impacts

Land Use. More than 50% of the United States 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 33 for a summary of land uses and population densities in several subbasins within the Columbia River watershed (data from Stanford et al. 2005).

Table 36. Land uses and population density in select tributaries of the Columbia River

Watershed	Land Use Categories (%)				Density
	Agriculture	Forest	Urban	Other	(people/mi^2)
Snake/Salmon rivers	30	10-15	1	54 scrub/rangeland/barren	39
Yakima River	16	36	1	47 shrub	80
Willamette River	19	68	5	--	171

Data from Stanford et al. 2005

The interior Columbia Basin has been altered substantially by humans causing dramatic changes and declines in 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, a variety of recreational activities, and urban uses. The decline of salmon runs in the Columbia River is attributed to loss of habitat, blocked migratory corridors, altered river flows, pollution, overharvest, 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 the 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, water milfoil, waterweed, cattail, and duckweed grow in the floodplain wetlands and shallow lakes. In the Yakima River, non-native grasses and other plant are commonly found along the lower reaches of the river (Stanford et al. 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 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 the creation of more irrigation canals and the passage of the Reclamation Act of 1902 (NRC 2004). Today, agriculture represents the largest water user within the basin (>90%). Agriculture, ranching, and 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 from 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 USGS 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 basin location 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 USGS between May 1999 and January 2000 detected 25 pesticide compounds (Ebbert and Embrey 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 a maximum of 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 aquatic mobile pesticides (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; Furher 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 ban 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 examined 11 species of fish, including anadromous and resident fish collected throughout the Columbia River Basin for a suite of 132 contaminants, including 51 semi-volatile chemicals, 26 pesticides, 18 metals, seven PCBs, 20 dioxins, and 10 furans. The study 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).

Urban and Industrial Development. The largest urban area in the basin is the greater Portland metropolitan area, located at the mouth of the Columbia River. Portland’s population exceeds 500,000, and the next largest cities, Spokane, Salem, Eugene, and Boise, have over 100,000 people (Hinck et al. 2004). Overall, the basin’s population density is one-third the national 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 in Hinck et al. 2004). According to Rosetta and Borys (1996 in Hinck et al. 2004), 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 (Rosetta and Borys 1996 in Hinck et al. 2004). The paper and allied products industry is the primary source of the suspended sediment load (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 significant pollutant loading to the lower basin, including most organics and over half of the metals. Although rural non-point sources contributions were not calculated, Rosetta and Borys (1996 in Hinck et al. 2004) surmised that in some areas and for some contaminants, rural areas may contribute a large portion of the load. This is particularly true for pesticide contamination in the upper river basin where agriculture is the predominant land use.

The Columbia River Estuary is under threat from several anthropogenic sources. Habitat loss has fragmented habitat and human density increase has created additional loads of pollutants and contaminants (EPA 2006). Water quality has been reduced by phosphorus loads and decreased water clarity, primarily along the lower and middle sections of the Columbia River Estuary. Although sediment quality is generally very good, benthic indices have not been established within the estuary, and fish tissue contaminant loads (PCBs, DDT, DDD, DDE, and mercury) are high, presenting a persistent and long lasting effect on estuary biology. Health advisories have been recently issued for people eating fish in the area that contain high levels of dioxins, PCBs, and pesticides. Habitat loss has been significant; 77% of swamps, 57% of marshes, and over 20% of tree cover has been lost to development and industry. Twenty-four threatened and endangered species occur in the estuary, some of whom are recovering and others (i.e., Chinook salmon) are not. Issues surrounding damming and environmental toxins have played key roles in original decline and subsequent recovery of several species and will be vital for future management. Invasive species in the estuary are pervasive; at least 81 have currently been identified, composing one-fifth of all species in some areas, and new species are being identified presently.

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 River 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 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 croplands and the Yakima Project delivers water to nearly 500,000 acres (BOR 2007).

The Bonneville Power Administration, an agency under the U.S. 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 20th 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, 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 1979). 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, estuary dynamics have 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 salmon production within the basin from the dams. 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 over 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 loss of genetic variability within and among populations (Busack 1990 *in* Hard et al. 1992; Riggs 1990; Reisenbichler 1997), disease transfer, increased competition for food, habitat, or mates, increased predation, altered migration, and displacement of natural fish (Steward and Bjornn 1990; Fresh 1997). 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). 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 a 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. This effort was mandated by Congress in 2005, and is in its early stages. 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.

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 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 (1979), 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 in the headwaters of the Snake River; the overall output from these deposits accounts for 12% of United States phosphate production (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 et al. 2005; EPA 2007). According to the United States 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 et al. 1997 *in* Hinck et al. 2004). Contaminants 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, which 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 riverboats 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 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 known harvest of Chinook salmon 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 in Mann et al. 2005). Recreational catch in both ocean and in-river fisheries varies from 140,000 to 150,000 individuals (Mann et al. 2005).

Puget Sound Region

Natural History

The Puget Sound watershed is defined by the crest lines of the Olympia Mountain Range (and the Olympic Peninsula) to the west and the Cascade Range to the east. The Olympic Mountains reach heights of about 8,000 feet, and are extremely rugged and steeply peaked with abrupt descents into the Puget Lowland. The Cascade Mountains range in heights of 4,000 to 8,000 feet with the highest peak, Mount Rainer, towering at 14,410 feet above sea level. As the second largest estuary in the United States, Puget Sound has about 1,330 miles of shoreline and 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 in the Central Basin at over 920 feet in depth. 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 and support habitat for invertebrates, fish, 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 in the San Juan Islands (Wydoski and Whitney 1979). Relief in the valley is moderate, with elevation ranging from sea level to about 1,300 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 Range to 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 the Puget lowland region – the lowland forest and the mid-montane forest of Pacific silver fir and Alaska yellow cedar; the subalpine forest of mountain hemlock with subalpine fir 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 to 35 inches, and falls predominantly as rain in lowland areas. Annual snowpack in the mountain ranges is often high; although the elevation of the Olympic 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. 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 1979; Kruckeberg 1991).

Major rivers draining to Puget Sound from the Cascade Mountains include the Skagit, Snohomish, Nooksack, Puyallup, and Green rivers, as well as 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 provide important salmonid habitats despite their small size.

The Puget Sound basin is home to more than 200 fish and 140 mammalian species. Salmonids within the region include coho, Chinook, sockeye, chum, and pink salmon, kokanee, steelhead, rainbow, cutthroat, and bull trout, as well as Dolly Varden (Wydoski and Whitney 1979; Kruckeberg 1991). Important commercial fishes include the five Pacific salmon and several rockfish species. A number of introduced species occur within the region, including brown and brook trout (*Salvelinus fontinali*s), Atlantic salmon, bass, tunicates (sea squirts), and a saltmarsh grass (*Spartina* spp.). 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 Their 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" forestland and by 2004 the total acreage was nearly half that originally surveyed (PSAT 2007). 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 occurred in the Urban Growth Boundary, which encompasses specific parts of the Puget Lowland; in this area forest cover declined by 11% between 1991 and 1999 (Ruckelshaus and McClure 2007). Projected land cover changes (reviewed in 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 2007) 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 watersheds in the region, increased about 16% 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 1996). 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 result 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 a concrete diversion at river mile 8.5, resulting in a permanent increase of Puyallup River flow 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. Over time, the Duwamish 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 roughly 18% 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%.

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 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 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 2007) 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) of submerged lands within Puget Sound are considered moderately contaminated. 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 (PSAT 2007). Hypoxic or low dissolved oxygen concentration were found at a number of monitoring stations, including Saratoga Passage, Discovery Bay, Bellingham Bay, Elliott Bay, Budd Inlet, and Commencement Bay. Many of the contaminants in the Sound, including several that were banned years ago, continue to bioaccumulate in the food web to top level predators (NMFS 2008a).

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 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, bank hardening for erosion control, and agriculture uses. Since the first dike on 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, allow for floodplain activities (like agriculture or development), and to power local timber mills (Ruckelshaus and McClure 2007).

In the next couple of years, 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 suggest that nearly 400,000 salmon could begin using the basin within 30 years after the dams are removed (PSAT 2007).

About 800 miles of Puget Sound's shorelines are hardened or dredged (PSAT 2004). The area most intensely modified is the urban corridor – the eastern shores of Puget Sound 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 Washington, and in 2004 the state was ranked 13th nationally in total nonfuel mineral production value and 17th in coal production (Palmisano et al. 1993; NMA 2007). Metal mining for all metals (zinc, copper, lead, silver, and gold) peaked between 1940 and 1970 (Palmisano et al. 1993). 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 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 the 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, or docks. Entanglement of marine mammals in fishing gear is not uncommon and can lead to mortality or serious injury.

Oregon-Washington-Northern California Coastal Drainages

Natural History

This region encompasses drainages originating in the Klamath Mountains, the Oregon Coast Mountains and the Olympic Mountains, all of which form 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, Yaquima, 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 square miles and is a slightly over 110 miles long, and the Nehalem River drains a basin of 855 square miles and is almost 120 miles long. However, systems here represent some of the most biologically diverse basins in the Pacific Northwest (Johnson 1999; Carter and Resh 2005).

The region is part of a coastal, temperate rainforest system, and is characterized by a 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, 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, as well as 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. 1993). Approximately 92% of the Nehalem River basin is forested, with only 4% considered agricultural (Johnson 1999). Similarly, in the Umpqua River basin, about 86% is forested land, 5% agriculture, and 0.5% are considered urban lands. Roughly half the basin is under Federal management (Carter and Resh 2005).

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. (1993) dams in the coastal streams of Washington permanently block only about 30 miles of salmon habitat. 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, since effects of the channel scouring and loss of channel complexity resulted in the long term loss of salmon habitat (NRC 1996).

Mining. Oregon is ranked 35th nationally in total nonfuel mineral production value in 2004, while Washington was ranked 13th nationally in total non-fuel mineral production value and 17th in coal production (Palmisano et al. 1993; 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. 1993). 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 the 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 commercial landings in the region are groundfish, Dungeness crab, shrimp, and salmon. Many of the same species are sought by tribal fisheries, as well as 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, or docks.

Impact of the Environmental Baseline on Listed Resources

In 2007, the population of the United States increased to more than 300 million people for the first time in its history. That population growth and increase in population density was accompanied by dramatic changes in the landscapes of the United States. By 2000, half of the population in the United States lived in the suburbs (Hobbs and Stoops 2002). About 75% of all Americans now live in areas that are urban or suburban in character; that is, about 75% of the people in the lower 48 States live in less than 2% of the land area of the lower 48 states. Most modern metropolitan areas encompass a mosaic of different land covers and uses (Hart 1991). The mosaic or land uses associated with urban and suburban centers has been cited as the primary cause of declining environmental conditions in the United States (Flather et al. 1998) and other areas of the world (Houghton 1994).

The direct and indirect effects of these changes in land-use and land-cover have had lasting effects on the quantity, quality, and distribution of every major terrestrial, aquatic, and coastal ecosystem in the United States, its territories, and possessions. Many native ecosystems exist as small isolated fragments, surrounded by expanses of urban and suburban landscapes or by natural areas dominated by non-native species. As a result, many of the native plant and animal species that inhabited those native ecosystems over the past have become extinct, endangered, or threatened over the past 200 years. Even marine ecosystems, once deemed by many as the most resilient of ecosystems, a vast source of fish for harvest and a limitless sink for waste material, are threatened by human activities on a global scale. The most pervasive threats to marine ecosystems include ocean-based destructive demersal fishing practices, increasing sea temperatures, coastal development, increased sediment loading, point-source organic pollution, and hypoxia (Halpern et al. 2007).

The rapid growth of commercial fishing of what was once considered an endless food supply has resulted in drastic over-exploitation of fisheries resources and modification of the marine environment (Hall 1999). Increases in national and global populations have lead to a dramatic increase in demand for seafood, resulting in expansion of fishing fleets by orders of magnitude, development of new technology to capture resources more efficiently, and greater ability to exploit areas once considered out of reach. In particular, fishing practices have lead to pressures not only on target species, but changes to whole habitats and the protected species that are either caught directly, or whose habitat is degraded because of them. It has been estimated that global commercial fishing industries catch and discard 27 million metric tons of fish, sea turtles, marine mammals, and other organisms annually (Hall 1999). Gill nets set for several miles can entrap, drown, or disable any organisms larger than their mesh size, from salmon to large whales. Although gill nets may be set thousands of miles from domestic waters, individuals of protected species caught in these nets can be the same that nest, breed, or feed in United States waters.

Dredging and trawling gears clear bottom habitat of any sizeable material, eliminating habitat of small fishes and invertebrates on which other species feed (Hall 1999). This process also displaces large amounts of sediment into the water, dramatically altering water clarity and chemistry. There are likely additional factors that influence listed species directly or indirectly, which are thus far unknown.

The process of global warming is a developing concern to protected species management. Widespread habitat alteration or loss can also stem from even moderate, but prolonged, increases in temperature. Although many effects of climate change are unknown, the instability and environmental change that has been measured to occur thus far support the likelihood that global warming will have negative impacts on protected species and the habitats that they occupy.

Coastal development has more localized effects on marine environments, but is so extensive that most, if not all, nearshore environments are affected by it in some way. Development may be so detrimental as to extirpate populations or species in very short periods of time. Such is the case with several populations of salmon along the United States Pacific coast, where dam construction blocked fish movement to and from spawning and feeding habitats (Lichatowich 1999). As a result, entire populations are now considered extinct. In general, coastal development without environmental consideration has resulted in direct mortality to protected species, modification of habitat to displace individuals or populations from a region, or reduced reproductive success. In such cases, survivorship declines can be significant, resulting in protection of species not formerly listed, or moderate in species already listed that can ill-afford further impediments to recovery. As with fishing, coastal development in foreign countries can affect marine species protected in the United States by affecting habitat to which these species migrate for breeding or feeding. Environmental impacts, particularly to strategically important or listed species, of coastal development have received more global interest in recent years and changes, such as EIS statements, outreach and education, and environmentally friendly design have mitigated some impacts. However, many countries continue developing coastal regions without significant concern for protected or sensitive species or their habitats and these distant activities can have negative consequences for listed species in this country.

Additional activities on land have significant effects in ocean environments. This is particularly true for sedimentation as well as agricultural, industrial, and municipal pollution. Soils are normally covered by tracts of forest, grassland, marsh, or other vegetation preventing significant erosion. However, development activities tend to disturb these areas, or bring in large amounts of soil during construction, allowing for wind, rain, and other mechanisms to move the soil to local water bodies. Salmon nests become covered with sediment, or highly localized spaces for nests become covered, resulting in high hatching mortality or elimination of entire stretches of spawning habitat (NMFS and USFWS 2005).

Agricultural development and use has its own unique contribution to marine pollution. Fertilizers applied to tracts of land, from front lawns to large fields, can run-off in rainwater if not applied properly these fertilizers contain concentrated nutrients that dissolve in water and enter streams, rivers, lakes, estuaries, and the marine environment (Kennish 1992; Soares 1999). Along with nutrients contained in sediments, these elevated nutrient concentrations provide fodder for potentially exponential bacterial, algal, and plant growth. This rapid growth process can create algal “blooms” (red tide), which can make toxic metabolic byproducts in such concentrations that fish, seabirds, and marine mammals can become ill or die as a result. Such events happen continually along Gulf of Mexico states and instances are known for the west and east coasts. After nutrients have been used up, large numbers of small organisms die and the natural breakdown of their bodies results in areas of oxygen depleted water, sometimes hundreds of square miles in size, called “dead zones” in which organisms requiring oxygen in water to breathe cannot survive. Such an area occurs off the coast of Louisiana. This process of eutrophication can eliminate large areas of nearshore and oceanic habitat, resulting in direct mortality to or adverse modification of habitat utilized by listed species. Shortnose sturgeon are generally believed to be absent from numerous rivers feeding into and sections of the Chesapeake Bay itself because of eutrophication issues stemming from fertilizer use on lawns and fields. Unlike most other forms of pollution, eutrophication can eliminate or displace large sections of habitat and all animals within it. This issue has received more interest in recent years. Regulations are being installed to regulate fertilizer runoff and public outreach has been growing.

Although sedimentation and agricultural pollution comes from general areas, point-source pollution comes from specific effluents and can have additional effects. These drainages frequently come from municipal wastewater treatment plants, commercial and industrial discharges, as well as recreational and commercial vessels (Kennish 1992). Point-sources tend to contain specific chemical components that result from anthropogenic activity, as opposed to excessive sediments entering a waterbody. These components can be toxic and require regulation. However, the effects of components on species and their environment is generally unknown and it is only after several years of research that enough evidence is collected to initiate regulation. Such has been the case with pesticides, such as DDT and DDE, which caused severe fragility in bird eggs and led to the listing of several avian species, including bald eagles. Such is now the case with pharmaceuticals in wastewater. Hormones are currently released in wastewater from treatment plants. It is unknown what effects these chemicals have on endocrine disruption to species in habitats near wastewater discharges. It has been suggested that humans reconsuming these waters may have intra-sex children (Soares 1999), which indicates that these chemicals may affect other exposed organisms. What is known is that point-source discharges can introduce chemicals into fresh water, estuarine, and marine habitats whose effects can cause significant decline in a variety species, but the effects may not be known for years later.

Salmonids originally underwent dual pressures that led to their decline: dam construction and commercial fishing (Lichatowich 1999). Although fishing had occurred extensively through time, more widespread and technologically advanced methods were developed in the past two centuries to harvest salmon beyond the rate at which they could reproduce (Lichatowich 1999). More importantly and at the same time, dam construction occurred that cut the connection between two necessary salmon habitats: streams and ocean (Lichatowich 1999). This lead almost immediately to large-scale salmon declines or extinctions in several local areas. Now, dams have generally been modified or removed to re-establish communication between habitats for salmon in most areas. Commercial fishing is also closely monitored to prevent excessive pressures on populations. However, new threats in the forms of habitat loss, pollution, and genetic dilution of populations specialized for certain habitats impede recovery efforts (Reisenbichler 1997). As predators, salmon tend to bioaccumulate toxins as whales do, but generally accumulate more because they eat other fish instead of krill, which are lower on the food chain. Pollution is identified as a contributing factor for 38% of ESA listed species overall (Hoffman et al. 2003). Contaminants can cause reproductive disruption, immune dysfunction, and other physiological effects accumulate in vertebrates and can cause reduced reproductive fitness and subsequent population decline (Rand and Petrocelli 1985).

EFFECTS of the Action

The *Description of the Proposed Action* describes EPA's proposal to continue to recommend the 1985 304(a) aquatic life criteria for cyanide and approve state and tribal water quality standards, or federal water quality standards promulgated by EPA for the protection of aquatic life that are identical to or are more stringent than the section 304(a) cyanide aquatic life criteria. The *Status of the Species and Critical Habitat* section of this Opinion identified the endangered and threatened species, and designated critical habitat that may be affected by the proposed action, as well as those species and critical habitat that are currently proposed for listing under the ESA. The *Status* also summarized the status and trends of those species, and other ecological information relevant to our effect's analyses, while the *Environmental Baseline* summarized the consequences of a variety of human activities, including land and water uses that impact the listed species and critical habitat considered herein.

In this section, we identify specific stressors and subsidies associated with the proposed action, the likelihood endangered species, threatened species and designated critical habitat are exposed to those stressors and subsidies, the responses of listed species and critical habitat to their exposure, and the consequences of those responses to the different listed resources. Based on the results of these analyses, we assess the risks EPA's proposal to recommend and approve of water quality standards for cyanide poses to listed resources. For endangered and threatened species, our assessment focuses on the risk of increasing the extinction probability of these species, for designated critical habitat our assessment focuses on the risk of reducing the conservation value of the habitat designated for the endangered and threatened species.

As discussed in the *Approach to the Assessment* section of this Opinion, as part of this national consultation, our consultation examines the decision-making process that EPA uses to recommend and approve water quality criteria and the outcome of that decision making process. In particular, this consultation focuses on how EPA determines what constitutes a "safe and healthful level in waterbodies for a pollutant, which a regulatory authority can use to guide the control, reduction and eventual elimination of that pollutant (BE, page 15)" in the environment for the protection of fish and wildlife species consistent with the goals of the CWA, and threatened and endangered species in particular.

When EPA recommends 304(a) aquatic life criteria, that recommendation means that water quality standards identical to or more stringent than EPA's criteria will protect the designated uses of water that receive pollutants at levels consistent with the aquatic life criteria. As a default, EPA uses "fishable and swimmable" as the designated uses when it establishes their aquatic life criteria. That is, EPA has determined that the adoption of their criteria to be protective of aquatic life designated uses consistent with the objective and goals articulated in

CWA sections 101(a) and 101(a)(2) (EPA 2008b). Therefore, when EPA recommended 304(a) aquatic life criteria for cyanide, EPA also determined that cyanide at the recommended numeric value would protect designated uses consistent with the objective of the CWA to "restore and maintain the chemical, physical and biological integrity of the Nation's waters (CWA §101(a))," and the goal to provide "for the protection and propagation of fish, shellfish, and wildlife and provides for recreation in and on the water…. (CWA §101(a)(2))."

If EPA recommends 304(a) aquatic life criteria, then fish and wildlife that might be exposed to pollutants at those criteria levels generally should not experience physical, physiological, behavioral, or ecolgocal consequences that would interfere with reproduction or reduce the long-term persistence of their populations resulting from that exposure. That is, EPA expects that the criteria would generally provide a "reasonable level of protection" of all but a small fraction of the "appropriate" taxa (0.05; Stephan et al 1985). Restated, there is 5% probability that an aquatic species would not be protected by EPA's national criteria. Provided EPA considers threatened and endangered species part of the taxa that would be protected by their national criteria, then we would expect that EPA's national criteria would generally protect endangered and threatened species and designated critical habitat. Specifically, we would expect that when EPA recommends the CMC of 22.36 μg/L and the CCC of 5.221 μg/L in fresh water, or at the CMC of 1.015 μg/L or the CCC of 1.015 μg/L in salt water as its 304(a) aquatic life criteria for cyanide, then endangered or threatened species or designated critical habitat exposed to cyanide at these concentrations should not experience physical, physiological, behavioral, or ecological consequences that would reduce the long-term persistence of their populations resulting from that exposure. Certainly this would be the case if (a) EPA considered aquatic listed species as an indicator of or part of the aquatic assemblage that defines the biological integrity of the Nation's waters, or part of the fish, shellfish and wildlife the CWA intends to protect; or (b) listed species are expressly or indirectly listed as a designated use by a state or tribe.

We begin our assessment of the *Effects of the Action* by evaluating the decision making process EPA uses to develop 304(a) aquatic life criteria and establish numeric values for the CMC and CCC for a particular pollutant, and EPA's 1985 published values for the cyanide CMC and CCC. These values represent EPA's recommended 304(a) aquatic life criteria for cyanide, upon which EPA intends to subsequently approve for use in state or tribal water quality standards. Our evaluation focuses on whether it is reasonable to expect that endangered species, threatened species, and designated critical habitat are exposed to cyanide at concentrations similar to national criteria values; and whether it is reasonable to expect that endangered species, threatened species, and designated critical habitat are not likely to respond to any exposures to cyanide at the CMC of 22.36 μg/L or the CCC of 5.221 μg/L in fresh water, or at the CMC of 1.015 μg/L or the CCC of 1.015 μg/L in salt water.

If listed resources are likely to respond to exposures to cyanide at the CMC of 22.36 μg/L or the CCC of 5.221 μg/L in fresh water, or at the CMC of 1.015 μg/L or the CCC of 1.015 μg/L in salt water, then we would evaluate the likelihood that endangered species, threatened species, and designated critical habitat would be exposed to: a) the one-hour average exposure concentrations of cyanide that would not exceed the CMC more than once every three years; and b) four-day average exposure concentrations of cyanide that would not exceed the CCC more frequently than once every three years on average. If we conclude that, endangered species, threatened species, and designated critical habitat would be exposed to cyanide at concentrations that deviate from the one-hour and four-day average, we would examine the variability in concentrations to which endangered species, threatened species, and designated critical habitat would be exposed. As part of this evaluation, we would examine whether endangered species, threatened species, and designated critical habitat "should not be affected unacceptably (EPA 1985)" if the four-day average concentration of cyanide does not exceed 5.2 µg/CN in fresh water or 1.015 µg/CN in salt water more than once every three years on average and if the one-hour average concentration does not exceed 22.36 µg/CN in fresh water or 1.015 µg/CN in salt water more than once every three years on average. Finally, we would evaluate the context for probable exposure events including whether environmental conditions in which listed species reside or the physiological state of the individual organism would influence the severity of probable responses.

EPA's Decision-Making Process

<u>Derivation of Criteria</u>

In order to evaluate whether the cyanide aquatic life criteria and any water quality standards that would be based on those criteria are not likely to jeopardize listed species or adversely modify critical habitat, we first examine how EPA derived the aquatic life criteria. The EPA document, *Guidelines for Deriving Numerical National Water Quality Criteria for* the *Protection* of *Aquatic Organisms and Their Uses* (the "Guidelines") outlines the process that EPA uses to derive water quality recommendations that intend to protect aquatic assemblages (Stephan et al. 1985; EPA 2008b). According to the guidelines, once a decision is made that a criterion is needed EPA collects and reviews all available information on the toxicity of the chemical is collected and reviewed for acceptability, and sorted.

The decision-making process for deriving aquatic life criteria involves a mix of quantified estimates of the effects a particular contaminant would have on a sample of test subjects, and professional judgement. That is, criterion development involves quantifying the sensitivity of a suite of species to toxic compounds in controlled studies; professional judgment comes into the process in several areas including the setting aside of data, determining whether a species is commercially or recreationally important and whether data on that species deserves additional attention in the final derivation of the criterion, determining whether particular data is useful or should be set aside (e.g., determining if water quality characteristics of a test are acceptable, or whether the degree of agreement between species is reasonable).

As a general matter, the *Guidelines* require the use of acute and chronic toxicity tests on a broad range of aquatic species to provide an indication of the sensitivities of untested species. These data are used by EPA to develop chronic and acute criteria for both salt and fresh water, the CCC and the CMC respectively. EPA's development of two values for fresh and salt water, the CMC and CCC, is premised on the assumption that doing so more accurately reflects toxicological and practical realities while not being as restrictive as a one-number criterion would have to be in order provide the same degree of protection (Stephan et al. 1985).

To derive an acute criterion for fresh water, the *Guidelines* suggest that toxicity data be available for at least one species of freshwater animal in at least eight different families. The families include:

1) Salmonidae (e.g., salmon or trout),
2) a second family in the class Osteichthyes, preferably a commercially or recreationally important warmwater species (e.g., bass, bluegill),
3) a third family in the phylum Chordata (e.g, salamander, frog),
4) a planktonic crustacean (e.g, daphnia),
5) a benthic crustacean (e.g, crayfish, amphipod),
6) an insect (e.g., dragonfly, mayfly),
7) a family in a phylum other than Arthropoda or Chordata (e.g., mussel, snail, worm), and
8) a family in any order of insect or any phylum not already represented.

For deriving a saltwater acute criterion the *Guidelines* suggest that acute tests with at least one species of saltwater animal in at least eight different families should be used. The represented families should include:

1) two families in the phylum Chordata,
2) a family other than Arthropoda or Chordata,
3) either the Mysidae or Penaeidae family,
4) three other families not in the phylum Chordata,
5) and any other family.

Additionally, at least one acceptable test is required for saltwater and freshwater plants, and at least one acceptable bioconcentration factor determined with an appropriate saltwater species. Data that is rejected from further consideration may include: data from studies that did not contain control treatment; too many organisms in the control treatment died or showed signs of stress or disease; data from tests using distilled or deionized water as the dilution water without adding appropriate salts; data from species that do not have reproducing wild populations in North America; data on organisms that were previously exposed to substantial concentrations of the test material or other contaminants (Stephan et al. 1985).

Studies used for determining the CMC are acute tests, which are performed with 48 or 96 hours of exposure, and measure the concentration at which the toxin causes death in 50% of the test population (LC_{50}). The LC_{50} values for each species are pooled and averaged to determine the species mean acute value (SMAV). If EPA has data on several species within a genus, then the data are pooled again to calculate a genus mean acute value (GMAV). If data are available from only one species, then that species mean value becomes the GMAV. Once calculated, the GMAV is ranked from high to low (least to most sensitive species) and the lowest four values are used in regression to estimate the concentration that would cause death for the fifth percentile of the most sensitive species. This fifth percentile value represents the final acute value or FAV. In the event a commercially or recreationally important species has a SMAV below the FAV, the SMAV can be substituted for the FAV to protect that important species. Once EPA has determined the FAV (or the lowest SMAV for an important species) then that value is divided by two, in an effort to avoid the death of exposed organisms. The resulting value is the criterion maximum concentration or CMC. EPA calculates the CMC under the assumption, that the CMC averaging period would be substantially less than the lengths of the acute tests upon which it is based (Stephan et al. 1985). As such, EPA recommends that the CMC be applied as a limit on the 1-hour average concentration in the environment to provide an addition level of protection.

Chronic toxicity values are calculated either in the same general manner as the acute values, or by dividing the FAV by the final acute-to-chronic (ACR). The ACR is a way of relating the acute toxicities to chronic toxicities and is more commonly employed because it allows EPA to make use of a smaller data set. Chronic toxicity test data must be available from at least three different families, so long as at least one is a fish, an invertebrate, and one is an acutely sensitive species, in order to derive a final chronic value. In contrast to acute studies, chronic tests may last weeks or more, at sublethal exposure concentrations and focus on the endpoints of growth and reproduction. Chronic studies focus on two levels of effect for a concentration: the NOEC and the LOEC that cause a statistically significant change in the endpoint of interest (growth or reproduction). Similar to the FAV, the CCC is derived by pooling values and calculating the geometric mean of the two effect levels.

EPA’s decision-making process was developed under the assumptions that:

1) Effects that occur on a species in laboratory tests generally occur on the same species in comparable field situations (Stephan et al. 1985);
2) Effect levels defined by chronic toxicity tests are conducted on the “most sensitive life stages” and therefore should protect all other (less sensitive) life stages (Stephan et al. 1985)
3) When the minimum data requirements are satisfied, but few data are available, then restrictive criteria values are derived (BE 2006).
4) The averaging recommendation is based in part on the assumption that most bodies of water could tolerate exceedences once every three years on the average provided the body of water is not subject to anthropogenic stress other than the exceedences of concern (Stephan et al. 1985).

Important caveats to the general approach in EPA’s decision-making process include:

1) The development of water quality standards may need to take into account additional factors such as hydrological considerations, environmental and analytical chemistry, extrapolation from lab to field situations, and relationships between species for which data are available and species in the water of concern (Stephan et al. 1985),
2) It may be necessary to derive site-specific criteria by modifying national criteria to reflect local conditions of water quality, temperature, or ecologically important species.
3) Some untested locally important species might be very sensitive to the contaminant of concern (Stephan et al. 1985),
4) Some aquatic organisms in the wild may be stressed by disease, parasites, predators, other pollutants, contaminated or insufficient food, and fluctuating and extreme conditions of flow, water quality and temperatures (Stephan et al. 1985),
5) The decision-making approach is meant to derive criterion that prevent unacceptable long-term and short-term effects, which is not the same as threshold of adverse effects. Some adverse effects (e.g., small reductions in growth, survival or reproduction) will probably occur at or below criterion values (Stephan et al. 1985),
6) The frequency, magnitude and duration of the exceedences should be based on the ability

of the aquatic ecosystem to recover, which will differ greatly according to the pollutant and the state or health of the ecosystem (Stephan et al. 1985)

Understanding the assumptions and the caveats inherent to EPA's decision-making process is important to understanding the uncertainty around the values EPA recommends to states and tribes for use as their water quality standards. For instance, according to EPA laboratory tests conducted at constant exposures simulates "worst case" field conditions. In limited circumstances this assertion is probably true, but in many cases it is not. In the wild, species will typically not be exposed to continuous concentrations of a particular chemical. Rather, concentrations typically vary temporally and spatially and would result in doses that are both higher and lower than the tested dose. This in itself does not make the laboratory exposure approach a reasonable simulation of a worst-case field (or natural) condition. Responses of organisms tested in controlled laboratory systems do not necessarily provide reasonable predictors of organisms' responses to similar chemicals in the wild, although admittedly in many cases this is the only type of data available to us from which to conduct an evaluation. In many cases, the conditions simulated in a laboratory test have little to do with the environment in which most species live in the wild, and as such are unlikely to resemble "worst case field conditions."

In laboratory tests, species are generally isolated from multiple stressors so that researchers are able to isolate the species responses to the chemical (or stressor) under study. In the wild, species are typically exposed to a wide range of stressors, from natural to human induced. For instance, lab studies do not replicate typical environmental conditions where intraspecific competition for food or shelter occurs. Instead, all the test organisms are about the same size, provided with abundant food, and minimal habitat complexity. Interspecific competition generally does not occur in lab tests either, as most lab environments isolate the species under study from typical predators. Physical conditions are maintained at optimal or constant levels (e.g., velocities, water temperature, and dissolved oxygen are not representative of fluctuating conditions in a natural aquatic environment) and generally, there are no other chemical stressors present. Typically, lab specimens are generally not exposed to other stressors such other chemicals, or environmental factors that can influence toxicity (e.g., some chemical or environmental changes in temperature or other parameters can increase or decrease toxicity, some times in a greater than additive fashion). Wild taxa are exposed to a myriad of factors that can influence their responses to a particular chemical at a particular concentration, and at best the laboratory tests are an indication of how species may respond to that chemical in nature. The actual physical and chemical conditions within a waterbody can, for some chemicals, alter the toxicity of the chemical evaluated in the laboratory under controlled conditions. Knowing this, the authors of EPA's decision-making process noted that it may be necessary to account for local conditions when setting water quality standards and permit limitations (see caveat 1 above).

Another important assertion is that EPA's decision-making process uses the most sensitive life stages for defining chronic toxicity. Unfortunately, chronic values, as is the case of acute values, are calculated on available data and generally, chronic studies are few in comparison to studies that examine mortality as the endpoint of concern. The species used for lab tests are also often not representative of the composition and sensitivities of species in a natural community or ecosystem. EPA's aquatic life criteria guidelines require species from eight different families be tested to determine acute toxicity values for both marine and fresh water. To derive chronic numeric criteria, however, only three chronic tests are necessary, despite the fact that chemical concentrations in the natural environment are likely to occur more often at chronic low levels. Use of such a small data set to make inferences to a much larger community in the wild is cause for concern. Further, it is unclear whether the assumption that the most sensitive life stage is tested, is regularly met. Certain life stages or the transition between life stages, which are inherently stressful as a result of the physiological changes the animal is undergoing (e.g., osmoregulation), are rarely tested.

Even if the tests are conducted on the most sensitive life stage to a particular toxicant, it does not necessarily follow that the critical concentration determined by these sensitive stages are correlated with the vulnerability of the species to the toxicant. For instance, Kammenga et al. 1996 demonstrated that the fitness implication of a toxicant was measureable on the least sensitive stage of the tested species, whereas the most sensitive trait did not have any effect on the fitness of organism. Equally important, however, is that the smaller the data set used to extrapolate responses, the lower the confidence can be in the outcome of the final value. As is the case for many compounds, for cyanide the most robust data set that EPA had available for making its decision was the acute data set. Only five studies were used for deriving the CCC for fresh water, and two from saltwater taxa ---none of which represent empirical evidence of how any of the species addressed in this Opinion would respond to low-level or prolonged exposures of cyanide.

According to EPA when the minimum data requirements are satisfied, however, then *restrictive* criteria values are derived. Unfortunately, extrapolating the stress responses of individuals in a limited number of lab tests to organisms exposed to similar chemical concentrations while in highly complex natural environment provides for weak gross scale predictions at best, particularly when few or none of the species of interest were evaluated by direct empirical evidence. The greatest utility in simple laboratory tests is that they facilitate faster (and cheaper) data on generalized responses of a range of taxa to a defined chemical exposure (Cairns and Niederlehner 2003). Models, mesocosm or field studies, transparent reasoning, and validation studies should temper the results of such lab studies in decision-making, particularly when extrapolating potential environmental outcomes to a complex environment and in situations, like the management of threatened and endangered species and their designated critical habitat, where a low tolerance for error is warranted. Stephan (2002) summed it up best when he said: “Unless species are selected from a field population using an appropriate procedure (e.g., using random or stratified random sampling), use of the resulting benchmark(s) to protect field populations requires a leap of faith that the distribution of the sensitivities of tested species is representative of the distribution of the sensitivities of field species.”

Consideration of Listed Resources in EPA’s Decision-Making Process

EPA’s decision-making process (a.k.a. the *Guidelines*) does not explicitly require EPA to consider toxicity data on endangered or threatened species, although one species in particular, *Oncorhynchus mykiss* (specifically the freshwater phenotype, rainbow trout) is a commonly tested fish species. How EPA incorporates threatened and endangered species into their approval of state and tribal water quality standards varies across regions. Certain regional offices of EPA have completed Section 7 consultations on their approval of state water quality standards for a subset of the numeric standards. This national consultation represents the first of a series of Section 7 consultations with EPA on their recommended criteria and EPA's subsequent approval of state and tribal water quality standards that are based on the recommended 304(a) aquatic life criteria. This enhanced coordination at the national level was envisioned under an MOA between the Services and EPA (66 FR 11202).

There is a critical difference between decision-making for the purpose of criteria setting and conducting a risk assessment on a particular species or group of species (Suter and Cormier 2008). The benchmark calculation used in EPA's decision to recommend a particular criterion "rests on an assumption that selecting a percentile [e.g. 95%] is an appropriate way of specifying a level of protection (Stephan 2002)." Whereas, the Section 7 consultation solves (or attempts to solve) the risk of exposing listed species to a particular federal action or set of actions, in this case the risk of exposing listed species to chemicals at particular concentrations. Unlike criteria development, Section 7 consultations begin by assessing the effect of the chemical to the individual of a listed species. This endpoint differs greatly from the population level response evaluated during criteria development.

To bridge the gap between the aquatic-life criteria decision-making process and information needed to conduct Section 7 consultation, EPA with the assistance of the Services, developed the *Draft Framework for Conducting Biological Evaluations of Aquatic Life Criteria: Methods Manual*. The Methods Manual describes a process for evaluating whether the CMC protects acute mortality of listed species, and whether the CCC protects listed species under longer exposures. Additionally the Method introduces a process for evaluating the effects expected from a diet of aquatic organism contaminated with the chemical of interest to levels that would result from concentrations consistent with the criterion. The Method also addresses toxicity of the criterion chemical to the food items of listed species to determine if listed species are likely to be adversely affected by a loss of food. The basic goal of the Methods Manual was to produce robust decisions for determining when the aquatic life criteria for a specific chemical is likely to adversely affect (or not) a particular listed species, and whether formal consultation is required.

The Methods Manual Approach to Estimating Acute Responses. To evaluate whether a listed or proposed species would respond to a particular chemical when exposed at the criterion value, the Methods Manual uses a risk paradigm or risk ratio for conducting toxicity screening that is based on the numeric value represented by the CMC[9] as the "assessment exposure concentration" (represented by C_A), divided by the "assessment effects concentration" (EC_A).

$$R = C_A/EC_A$$

The EC_A is an estimate of the highest chemical concentration that EPA portends would cause an acceptable small adverse effect and for acute effects that estimate is derived when the mean acute

[9] The BE and the *Methods Manual* refer to this as the "maximum exposure concentrations allowed by the criteria", but this is deceptive as the approved standard allows that discharges may exceed the established value for the CMC under certain circumstances. See our discussion under *Concentrations of Cyanide in U.S. Waters*. Therefore, we note here that the C_A = CMC (or the CCC).

value divided by 2.27[10]. For acute toxicity, the small level of effect is EC_0 to EC_{10}. Under this simple paradigm when $C_A < EC_A$ then the chemical concentration established by the aquatic life criteria "is not likely to adversely affect" listed species. Conversely, when $C_A \geq EC_A$ then the chemical concentration established by the aquatic life criteria is considered "likely to adversely affect" listed species (see *Methods Manual*, page 9). This risk paradigm, defined by the risk ratio, forms the foundation of the each aquatic life criteria consultation.

For listed species for which acute data exist, the relationship is straightforward. Using the mean acute value calculated for rainbow trout or steelhead exposed to cyanide we illustrate the calculation. For example,

If the steelhead mean acute value = 44.73 µg CN/L,

Then $EC_A = 44.73/2.27 = 19.70$

And, $R = 22.36/19.70 = 1.14$[11]

Under this framework, a species with an $R < 1$ is not likely to suffer lethal consequences when exposed at the CMC, and a species with an $R \geq 1$ is more likely to suffer lethal consequences when exposed to the pollutant of concern at the CMC. Using this framework, the farther the species' R-value is away from 1, the more confidence there is in the determination that the species is (or is not) protected when exposed to cyanide at the CMC.

For listed and proposed species without species-specific data the EC_A is calculated using data from surrogate species. Since we do not have species-specific data for most listed species; most of the assessments will likely either estimate LC_{50}s for species using the Interspecies Correlation Estimations (ICE) model or Species Sensitivity Distributions (SSD). EPA developed the ICE model using taxonomic level information for endangered species. ICE models are based on regression analyses of LC_{50}s measured for a listed species to LC_{50}s measured for the same chemicals for commonly used surrogate species, preferably based on a minimum of five test chemicals. If surrogate species have been tested for the chemical of interest, but the listed species of interest have not, the relationships are used to estimate the LC_{50} for the chemical and species of interest. When an ICE model is not available for a listed species, then an ICE model for the genus or family is used. In this instance, each higher order ICE model must contain at least two species that represent the genus or family for it to be useful. Due to the uncertainty in the correlations, EPA stated in the *Methods Manual* that they intended to estimate the LC_{50} using the lower 95% confidence bound of the ICE. On the other hand, the SSD is calculated from several surrogate species within the same taxonomic unit as the species of interest, to define possible LC_{50}s for the species of interest. According the *Methods Manual*, to increase the confidence in protecting listed species the 5th percentile in this distribution will be used, such that

[10] Note that we previously refer to this divisor as 2. The actual factor is 2.27, the inverse of 0.44, and "is based on 219 acute toxicity tests which showed that the mean concentration lethal to 0-10 percent of the test population was 0.44 times the LC50 (43 FR 21506, 18 May 1978)." In practice, such as the 1985 CN criteria document, EPA uses 2, but in the *Methods Manual* and this consultation EPA chose to not round to the nearest whole number but to use the fractional component as it was originally published in the *Federal Register*.

[11] If the CMC and the EC_A had been calculated with the same divisor, either 2 or 2.27, then in this example the rainbow trout R value would equal 1 because the CMC for cyanide was set using the rainbow trout SMAV.

the actual toxicity for the listed species should be higher than the chemical concentration estimated. When an ICE model was available for the listed species, or within the genus of the species of interest, the ICE model was given preference over the SSD. The *Methods Manual* lists a six-step approach for deriving EC_A estimates using surrogate data given the data that are available for closely related surrogates.

Given the lack of empirical information on the effects of many toxics on listed and proposed species, the Services and EPA will have to estimate to the best of their ability the potential effect using information from other species. Clearly, the validity and robustness of this risk ratio approach as a conceptual framework depends upon the value calculated for the EC_A. That is, the strength of the value (or range of values) represented by the EC_A depends ultimately on the identification, assimilation, and interpretation of evidence (i.e., the use best available scientific and commercial data) used in its calculation, which we expect will for most consultation predominantly come from surrogate species.

The Methods Manual Approach to Estimating Chronic Responses. To evaluate whether a listed or proposed species would respond to a particular chronic exposure to a particular chemical, the Methods Manual uses the same risk paradigm as described previously. For chronic toxicity, we used the numeric value represented by the CCC as the as the “assessment exposure concentration” (represented by C_A), divided by the EC_A.

As with the acute EC_A, the chronic EC_A represents an estimate of the highest chemical concentration in water or food that would cause an acceptable small adverse effect. For chronic toxicity, the acceptably small level of effect is the NOEC. Studies on the chronic effects of cyanide on listed species are few, and the literature search conducted by EPA was for a wide variety of species that have been tested. For chronic toxicity, the EC_A is based on the acute toxicity to the listed species, and the acute to chronic ratio (ACR) of surrogate species. The ACR is calculated as follows:

$ACR = SS\ LC_{50}/SS\ NOEC$

Where: SS LC_{50} is the LC_{50} for the surrogate species

SS NOEC is the No Observable Effects Concentration for the surrogate species

EC_As are estimated using the following equation:

$EC_A = LS\ LC_{50}/\ ACR$

Where: LS LC_{50} is the LC_{50} for the listed species

So for example, if the fathead minnow SS LC_{50} = 138 μg CN/L,
And, the NOEC = 13 μg CN/L
Then, ACR = 10.6

The listed species LC_{50} is then divided by the ACR to derive an $EC_{A,}$ which is compared to the CCC. Using this framework, when the $C_A < EC_A$ then the chemical concentration established by the aquatic life criteria “is not likely to adversely affect” listed species. Conversely, when the $C_A \geq EC_A$ then the chemical concentration established by the aquatic life criteria is considered “likely to adversely affect” listed species (see the *Methods Manual*, page 9).

Once the analysis produces a $C_A \geq EC_A$ for a particular listed species and contaminant combination, the *Methods Manual* provides little insight on the next step in EPA’s evaluation. The *Methods Manual* merely states that when a particular chemical combination is classified as "likely to adversely affect" a particular listed species, these will require additional consideration and analysis to determine “under what circumstances risks are unacceptable (*Methods Manual*, page 9).” Unfortunately, the *Methods Manual* does not clarify for the reader what type or extent of “additional consideration and analysis” is necessary in such circumstances, nor does it provide a definition of when risks would be considered unacceptable (or acceptable). In contrast, the implementing regulations for Section 7 consultation state that “Each Federal agency shall review its actions at the earliest possible time to determine whether any action *may affect* listed species or critical habitat. If such a determination is made, formal consultation is required… (emphasis added; 50 CFR 402.14).”

Neither the implementing regulations for Section 7 consultation nor the ESA use the terminology “unacceptable” as a qualifier to describe effects to listed species. Therefore, it is unclear what EPA intended by this statement in the *Methods Manual* in terms of their Section 7(a)(2) consultations. An obvious unacceptable effect under Section 7 would be when an agency’s action is likely to jeopardize the continued existence of listed species, or destroy or adversely modify critical habitat. Arguably, a reduction in the fitness of an individual of a listed species may also be considered unacceptable. The term’s use is not defined under the ESA.

EPA’s *Guidelines* may provide some insight into what EPA considers an unacceptable effect (Stephan et al. 1985). According to the *Guidelines*, the “protection of aquatic organisms and their uses should be defined as the prevention of *unacceptable* long-term and short-term effects on (1) commercially, recreationally, and other important species and (2)(a) fish and benthic invertebrate assemblages in rivers and streams, and (b) fish, benethic invertebrate, and zooplankton assemblages in lakes, reservoirs, estuaries, and oceans (emphasis added; Stephan et al. 1985).” According to Stephan (1986) his use of the term “unacceptable” in EPA’s *Guidelines* was intentional because it allows for flexibility in determining the level of protection that a waterbody might receive and recognizes that such decisions are based on value judgements. When the validity of a criterion derived for a particular body of water is “based on an operational definition of ‘protection of aquatic organisms and their uses’ that take into account the practicalities of field monitoring and the concerns of the public” as suggested by EPA’s *Guidelines*, then what drives the decision as to what constitutes an unacceptable risk is the level of protection (or conversely, adverse effect) that a particular criterion would have on a particular state or tribe’s *designated uses* for their waters. The designated uses assigned to a particular waterbody by a state or tribe are explicit value-statements of what a particular state or tribe wants to protect their water resources for.

It follows that an unacceptable risk under EPA’s decision-making process is one that fails to protect the designated uses for a waterbody. This is also consistent with EPA’s review and approval of state standards. If EPA’s line of inquiry as established through the *Methods Manual* leads them to a "may affect" or more specifically, a "may affect, likely to adversely affect" and assuming that the only risk that would be considered unacceptable is if the critierion under review fails to protect designated uses, the question that remains is whether EPA generally considers endangered and threatened aquatic species (and aquatic dependent as defined by the *Methods Manual*) a designated use. If listed aquatic species, however, are not specifically identified as a designated use by a particular state or tribe, we would ask whether EPA, states, and tribes would generally protect listed aquatic species, as part of the broader definition to protect species that are defined as "important", part of the aquatic "assemblage", or "fish and wildlife." That is, would listed species fall into any of the categories identified by the *Guidelines*:

1. commercially, recreationally, and other important species, and
2. (a) fish and benthic invertebrate assemblages in rivers and streams, and
   (b) fish, benethic invertebrate, and zooplankton assemblages in lakes, reservoirs, estuaries, and oceans?

The third category, "fish and wildlife" comes not specifically from the guidelines but from the language adopted by many states to describe their designated uses. The answer to the question "does EPA consider threatened and endangered fish or benthic invertebrates part of any of these categories?" is critical to understanding EPA's decision-making process both pursuant to the *Guidelines* and the *Methods Manual*, and this consultation.

Designated Uses

The *Approach to the Assessment* section of this Opinion identified that EPA's approval of a state water quality standard involves more than merely establishing a numeric value for a particular chemical pollutant, but also requires a positive finding from EPA that a state has adopted uses that are consistent with the requirements of the CWA and that their proposed criteria protect those designated uses. Thus, state designated uses are an action interrelated to EPA's approval of any state standards that rely on EPA's recommended CMC and CCC values. When a state modifies EPA's criteria or proposes their own water quality criteria, then EPA must evaluate and find that the criteria protect a state's designated uses. When EPA recommends a criterion or promulgates a federal water quality standard, EPA states that it would generally find its criterion support the designated uses and the goals of the CWA: to restore and maintain the chemical, physical, and biological integrity of the Nation's water (objective of the CWA); and provide for the protection and propagation of fish, shellfish, and wildlife (the interim goal). Thus a state or tribe that has identified acceptable designated uses under the CWA can expect that if they adopt EPA's recommended water quality criteria that EPA would approve the standard. When EPA approves state or tribal water quality standards, that approval implies that those standards protect the designated uses of the state's waters when state waters are exposed to chemical pollutants at levels consistent with the criteria.

Whether a state's water quality standards actually protect the designated uses is unclear, and likely varies by circumstance (e.g., pollutant, state, and use). A designated use is a goal statement for a water body that reflects the social and political value of the water. Like numeric criteria, each state has discretion to set their own designated uses. As a minimal standard, the

CWA requires states adopt use designations consistent with the provisions of sections 101(a)(2) and 303(c)(2) of the CWA. Thus, a state must adopt uses that provide for the protection and propagation of fish, shellfish, and wildlife, and other uses such recreation, agriculture and industry. If a state designates a use that does not address the “fishable and swimmable” goal, the state must complete a use attainability analysis (UAA) that justifies why such uses are not feasible, and that the state is establishing the highest attainable use, instead. A state has the discretion to make their uses as restrictive or loose as they desire, as long as they meet the “fishable and swimmable” goals of the CWA. While the designated use is a qualitative value statement for a waterbody, a criterion represents a scientific determination as to whether a particular water body can, given an ambient concentration of a pollutant, can still support the designated use (Gaba 1983). However, the designated use, while written in qualitative form, should be as specific as possible so as to be measurable or have meaningful and measureable surrogate indicators of goal (designated use) attainment (NRC 2001). According to the Government Accountability Office (2002), many states recognized that the linkage between their designated uses and their ability to measure attainment (or failure to reach attainment) was missing and acknowledged that they needed evaluation criteria to determine whether designated uses are being protected that are measured by reasonably obtainable monitoring data.

Accordingly, part of the problem that GAO (2002) and National Research Council (NRC 2001) noted was that many states’ designated uses may be overly broad. Many states designated uses were established in the 1970s when they had only 180 days to do so. Consequently, many states adopted the very general goal of the CWA to provide for the protection and propagation of fish and wildlife (GAO 2002). According to the NRC (2001) the problem with such broadly defined designated uses is that broader the use designation and the weaker the linkage between the use and any measurable indicator of attainment, the greater uncertainty and higher likelihood of error in subsequent determinations of use attainment. We found that many of the coastal states and states that contain listed species under NMFS jurisdiction have updated their designated uses in the past ten years (Designated Use Table -see Appendix B). Currently, designated uses include such uses as fishing/harvest, propagation of fish, protection, natural state, viable populations, diversity, species richness, and species assemblages. In our review we found only a few specified that the use was for a native fish community, and a few that did not appear to have a designated use that included wildlife. We were also curious whether listed aquatic species are directly or indirectly protected as part of the designated uses coastal states had adopted.

We found only one state, California, and one territory, Puerto Rico that explicitly addressed threatened or endangered species as part of their designated use. California’s designated uses include a broad statement that the waters must support the survival and maintenance of aquatic species that are protected, and Puerto Rico’s designated uses note that endangered and threatened species are included as part of the broader category of desirable species (Table 34). Other states have revised their designated uses to incorporate the specific needs of certain threatened or endangered species (e.g., Oregon and Washington adopted designated uses for the protection of Pacific salmon). Washington’s designated uses explicitly denote the following categories of aquatic life uses: char spawning and rearing; core summer salmonid habitat; salmonid spawning; rearing and migration; salmonid rearing and migration only and several others (WAC 173-201A-200). Washington’s designated uses should provide additional protection for Washington’s native char, bull trout and Dolly Varden, and several species of Pacific salmon that are listed as threatened or endangered, as well as others that are not listed. This is likely an improvement over the more generalized goals of “for the protection and propagation of fish, shellfish, and wildlife” or “fishable”.

Table 37. State designated uses that explicitly address threatened and endangered species.

State	Designated Use Name	Designated Use Description	EPA Effective Date
CA Regions 1, 2, 3, 4, 5, 6, 7, 8, 9	Rare, Threatened, Or Endangered Species	Uses of water that support aquatic habitats necessary, at least in part, for the survival and successful maintenance of plant or animal species established under state or federal law as rare, threatened or endangered.	8/18/1994
PR	Class SB, SC, SD	Coastal waters and estuarine waters intended for use in primary and secondary contact recreation, and for propagation and preservation of desirable species, including threatened or endangered species.	6/26/2003

Careful consideration of the relationship between the value statement of use and the manner of evaluating attainment of the use is essential. When the relationship between the endpoint and the indictor is weak particular life stages of regionally important species and regional biota may be under-protected. Portions of the native aquatic community may be left unprotected by omission and unique life histories may be overlooked. For instance, Washington’s designated uses may generally protect spawning salmon, but are under protective of early or summer migrating adult salmon for water temperature where warm water temperatures may interfere with gamete development during the migration and holding of the early migrating spawners (T. Hooper, pers. comm., October 28, 2008). Additionally, broadly defined designated uses are difficult to translate into meaningful and measurable criteria for determining whether uses have been attained. The closer a designated use is linked to its indicator, the chance of falsely concluding that the designated uses are being attained, when they are not, decreases.

To address this problem the NRC (2001) recommended greater stratification of designated uses at the state level to provide a logical link between designated uses and attainment of that use (NRC 2001). Considering that the designated use is the description of the desired endpoint for a waterbody and the criterion is the measurable indicator for determining attainment, using a stratified designated use framework could allow state’s to measure ranges of attainment, create stronger linkages between endpoint and indicator, decrease decision risk, etc. The further the criterion for determining attainment is apart from the desired condition (the designated use) the greater chance for introducing (or magnifying) error into the decision-making process.

Figure 4 illustrates some examples of water quality criteria as the measurable indicator for attainment of designated uses in relationship to the desired endpoint, attainment of uses (after NRC 2001). The unnumbered square represents the designated use for the water (depicted by a value statement such as “fishable” or “swimmable”). Square 1, the furthest from the designated use, represents measures of pollutants at their source (end of pipe measurements). Square 2 represents the chemical criterion as the measure of the ambient water quality condition, but may also include non-chemical measures (criteria) for physical attributes of ambient water quality such as dissolved oxygen and temperature. Square 3 represents criteria that are associated with physical or biological sources of pollution, and might include such measures as flow timing, pattern, non-indigenous taxa, channel sinuosity, etc. Square 4 represents biological measures of ambient water quality condition, such as those represented by indexes of biological community.

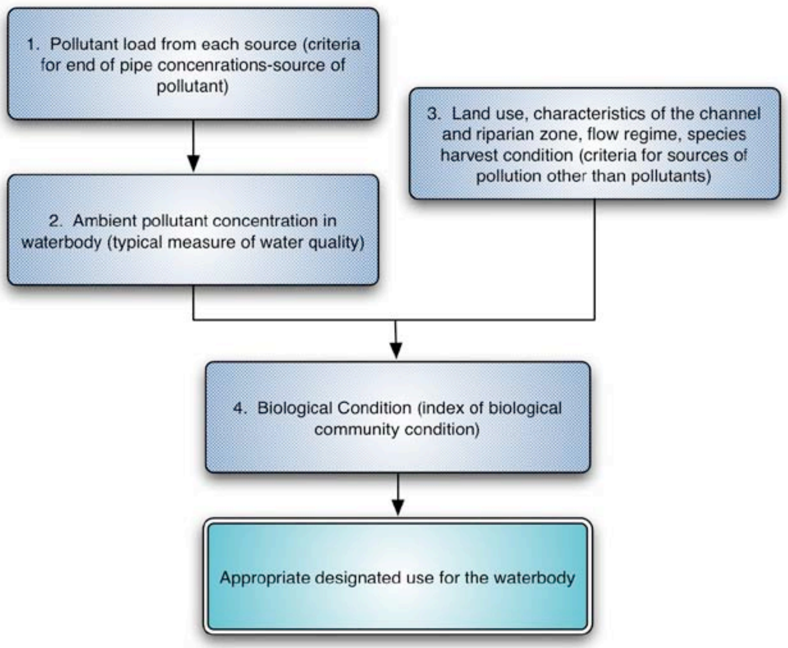

Figure 4. Types of water quality criteria and their position relative to designated uses (After NRC 2001).

A criterion, as described by NRC (2001) could be positioned at any point along the causal chain. However, if the desired endpoint is to restore and maintain the chemical, physical, and biological integrity of the Nation's waters, the biological condition is closest indicator to the desired endpoint. Not only is the proximate position of the biological indicator closer to the designated uses that describe the desired biological community, the biological community reflects the interplay between the physical, biological and chemical conditions of its environment. Under the stratified designated uses framework as suggested by the NRC (2001), states would adopt biological indicators as an intermediate and measurable indicator of designated use attainment. An index of biological health that considers a balanced community of native species versus the abundance and viability of alien species, loss of sensitive species and long-lived species; hydrological regime shifts (alterations in peak flows versus low flows, timing, intensity and duration), and so on, would provide a more holistic view of water body health and it's ability to meet public goals.

If the outcome or desired state for a designated use is preserving the biological integrity of the native community, then more meaningful measures as to whether that designated use is being supported by the aquatic life criteria are necessary. One advantage of a more explicit biological framing of designated uses is that threatened and endangered species can be expressly incorporated into the designated uses. When the designated uses are explicit, and provided the criteria properly support such designated uses, the broader biological community should be protected. In turn, it would be reasonable to expect that enhanced aquatic conditions may prevent more aquatic species from becoming listed under the ESA, and promote the survival and recovery of currently listed threatened and endangered species suffering from poor water quality. In contrast, when the biological community is not a measured indicator of what EPA intends to protect through its chemical indicators, then EPA and the states are engaged in a water quality process, including designation of uses, to "merely to justify the specific numbers contained in pollutant criteria (Gaba 1983)." Absent robust indicators, Gaba (1983) notes that EPA, in reviewing the adequacy of state water quality standards is also engaged in an "ad hoc" assessment of whether the states are satisfying the minimum requirements of the CWA, and what kinds of fish or other wildlife are to be protected under a particular designation (Gaba 1983; Stephan 1985).

NMFS is particularly concerned about those instances where EPA finds that a criterion can adversely affect certain populations of listed species, while simultaneously protecting designated uses. Although individual listed species and the population they represent are part of the native aquatic assemblage within a waterbody and depend upon quality waters for protection and propagation, according to EPA it cannot disapprove a state's designated use solely on the basis that the designated use does not provide for the protection against "take" of listed species (EPA 2008b). Yet, where a listed fish species is failing to mate, rear, feed, migrate, or maintain viable populations for reasons attributable, in part, to water quality, it follows that the standard is not providing for the protection and propagation of at least some fish.

States and tribes that wish to avoid water quality related impacts to listed species could write their designated uses to include the protection of listed species, as a general category and, if necessary, include species specific designated uses. When states include the protection of the viability of listed species as a designated use, as a general matter, those states should be able to demonstrate that they would not be likely to increase a listed species risk of extinction due to chemical water quality impacts so long as they are meeting their designated uses. To demonstrate this level of protection would require a strong linkage between the designated use and the criteria states use for evaluating attainment. States that rely on chemical criteria without biological criteria to measure the attainment of designated uses, and fail to designate biologically meaningful indicators of use, may miss important changes in environmental health attributable to water quality impacts, including changes in the viability of listed species populations (see for instance, Karr et al. 2003). Currently, however, the approach used by most states in evaluating the effectiveness of the criteria (and other water pollution control efforts) at meeting the designated uses is unlikely to present a very complete or comprehensive picture of the biological health of their waters from chemical or physical stressors, and therefore cannot provide a very complete picture as to the successfulness of the water quality control program (GAO 2002; Karr et al. 2003). According to Gaba (1983) EPA has allowed states to trivialize designated uses as a scientifically credible endpoint by allowing designated uses to justify the specific numbers contained in pollutant criteria, which EPA has predetermined support any designated uses that would comply with the very general goal of the CWA.

Arguably it is even more important that EPA recognize that confidence in the ability of aquatic life criteria to protect the aquatic assemblage is increased when chemical and biological criteria are used in concert to evaluate environmental impacts. The traditional laboratory based studies used as the basis for recommending aquatic life criteria require validation using more definitive and biologically rigorous metrics of biological integrity of natural systems. According to Adler et al. (1993, citing CRS, 1972 Legislative History, 76-77), the definition of “biological integrity” includes a condition in which the natural structure and function of ecosystems is maintained, and natural levels of biological integrity are those “levels believed to have existed before irreversible perturbations caused by man’s activities.” While the Senate report instructed that integrity under the CWA ought to be determined by reference to historical records on species composition (Adler et al. 1993). Biological integrity as defined by Karr and Dudley (1985) is “the capability of supporting and maintain a balanced, integrated, adaptive community of organisms having a species composition, diversity, and functional organization comparable to that of natural habitats of the region”. If it were EPA’s intent to design aquatic life criteria that protect the designated use for “fishable” waters, they would test the validity of *whether criteria are protecting the aquatic assemblage* in a waterbody, using rigorous biological indicators of aquatic ecosystem health.

The *native* aquatic assemblage is, arguably, the relevant endpoint envisioned by the Congress in establishing the CWA – when they stated the objective of the CWA is to restore and maintain the chemical, physical, and biological integrity of the Nation’s waters -- and is certainly is that envisioned by Congress in adopting the ESA. Regardless, EPA could engage states in better defining the objectives of their uses classifications, and identifying measureable indicators of attainment. More importantly, EPA should review their operational definition of protecting the aquatic assemblage in a waterbody and how the definition should be expanded beyond the limited indicators of species richness and species evenness to better reflect current science for a biological healthy aquatic community, and incorporate their affirmative duties under both the CWA and ESA (EPA 2008b). Species richness and species evenness are not necessarily indicators of the health of the native aquatic fauna. They can, however, be combined with other important variables for assessing the biological condition of a water body such as: species diversity, trophic composition, fish abundance, fish health metrics (e.g., body condition), presence (or absence) of non-native species, presence (or absence) of tolerant (or sensitive) species. EPA might also adopt a stratified designated use approach, rigorous and measurable indicators of the native aquatic assemblage in those states where EPA retains primacy for setting water quality standards, and engage in meaningful field studies for assessing the status of surface water integrity that integrates chemical criteria with indicators of biological and physical condition. While EPA has stated the guidelines for establishing aquatic life criteria are meant to protect aquatic assemblages, the laboratory studies used in setting the aquatic life criteria for cyanide do not represent species’ compositions from a natural community or ecosystem and consequently may fail to identify toxicant/population/community interrelationships. If field monitoring is not feasible, then mesocosm studies could provide EPA an opportunity to take a replicatable, laboratory-controlled approach to evaluate higher order effects in aquatic systems. Such studies may be useful in examining the indirect effects of reduced water quality and community response.

Stressors and Subsidies Associated with the Proposed Action

The primary stressor associated with the proposed action is aqueous cyanide. The following sections provide background on the characteristics of cyanide as a pollutant; including its uses and sources, observed concentrations, and other information that helps establish the exposure profile---the magnitude and spatial and temporal patterns of cyanide occurrence in the environment to which listed resources are exposed---for this analysis.

Exposure Analysis

Cyanide Sources and Production

We examined the typical sources of cyanide and the geographic distribution of those sources of cyanide to determine whether we would expect cyanide would co-occur with listed resources. This effort was based on the presumption that the fewer sources of cyanide there are across the United States, and the more limited their spatial distribution, the less likely that listed resources would be exposed to cyanide during their lifetime. If through this examination we would find that cyanide does not co-occur with listed resources, then we would conclude there is no exposure. The evidence leads us to conclude, however, that this is not the case. That is, based on the large number of sources of cyanide, their wide spatial distribution, and the increasing production of cyanide in the United States, we expect listed resources are more likely than not to be exposed to one or more sources of cyanide during their lifetime.

A common misconception is that cyanide is predominantly associated with gold mining or other mineral processing operations, which would tend to make this predominantly fresh water and perhaps rural pollutant. While cyanide is widely used in ore-extraction and cyanide related mine accidents have been widely publicized particularly when they have led to massive fish kills and human impacts, cyanide enters waterways from a wide variety of sources. Cyanide is ubiquitous in the environment, at least at low levels, as it is produced by a number of plants and microorganisms. However, cyanide is also produced synthetically to support industrial uses and is a byproduct of certain industrial processes (Leduc 1984; Eisler 1991; Dzombak et al. 2006).

Humans contribute the vast majority of cyanide to the environment. Cyanides are used widely in steel and heavy metal industries (e.g. electroplating), the manufacture of synthetic fabrics and plastics, as a pesticide and as an intermediate ingredient in herbicides, in road salts, and some fire retardants. Cyanide is also a byproduct of other activities such as municipal waste and sludge incineration and coking and gasification of coal (see Table 35). Of these sources metal industries and organic chemical industries are major contributors of cyanide into the freshwater aquatic environment, whereas, atmospheric cyanide, a by-product of forests fires, may be the primary source of oceanic cyanide except where cyanide enters coastal waters from fresh water sources (Leduc 1984; EPA 2005; Dzombak et al. 2006). Wastewater treatment plants across the United States can also be unexpected, but significant sources of cyanide to both fresh water and saltwater environments through several chemical processes, including dissociation of thiocyanide by chlorination or UV disinfection, chlorination in the presence of residual ammonia, nitrosation, and photolysis of ferrocyanate (Kavanaugh et al. 2003).

According to the 2002 United States Economic Census, there are 180 facilities engaged in gold-ore mining in 27 states across the nation, including Alaska, California, Idaho, Massachusetts, and Florida (U.S. Census Bureau 2002). The top four states, in terms of number of facilities, were Nevada, Colorado, California, and Alaska. In contrast, the manufacturing of photographic film, paper, plate, and associated chemicals occurs in more than 400 facilities and 24 states across the nation, and more than 3,000 establishments engage in electroplating and related activities in 41 states across the nation (U.S. Census Bureau 2002). The influx of cyanide to aquatic environments is likely as widely distributed across the landscape as the industries that use cyanide as part of their routine operations.

Cyanide is also synthetically produced in several states across the nation including Texas, Wyoming, West Virginia, Nevada, and Ohio (CMR 2008). In fact, the synthetic production of cyanide in the United States is a growing industry. The United States production of hydrogen cyanide (HCN) more than doubled in the past two decades from 330,000 tons in 1983 to 750,000 tons in 2001. Production growth between 1997 and 2000 increased about 1.7% per year (Dzombak et al. 2006; CMR 2008). The Chemical Market Reporter indicated that production demand in 2004 was estimated at nearly 2 million pounds. With demand exceeding current production of HCN, and price growth positive for the producers, HCN production and availability is expected to continue to increase in the United States. Incidentally, the United States does not export domestically produced HCN (CMR 2008).

The largest portion of the HCN produced in the United States is used in the textiles industry, for nylon production (47% is used for adiponitrile). Whereas, 27% is used in the production of acetone cyanohydrin for methyl methacrylate, the monomer for the transparent plastic polymethyl methacrylate also known as acrylic, 8% is for the production of sodium cyanide (NaCN), 6% is for methionine, 2% are chelating agents, 2% for cyanuric chloride, and 8% goes to miscellaneous uses including nitrilotriacetic acid and salts (CMR 2008). The demand for nylon remains high, with new growth and new applications still strong. According to CMR (2008) one such new application is in the automobile industry where metal components are being replaced by nylon parts. At the same time acrylic demands remain high, while the declining price of gold has reduced the demand for NaCN production, which had formerly been the primary driver for HCN production. With overall demand for HCN production growing in the United States, clearly cyanide is not a chemical that is being phased out of production or practical use but remains in prominent use. In fact, acrylonitrile (vinyl cyanide), a monomer in the synthesis of adiponitrile, is among the top 50 chemicals produced in the United States (Dzombak et al. 2006). While HCN facilities that support acrylonitrile production are in several states across the United States, several of the largest producers are in Texas (CMR 2008).

Table 38. Industrial Sources and Uses of Cyanide Compounds.

Source/Use	Form	Reference
Energy Production - Coal Gasification	Cyanide salts (potassium cyanide, sodium cyanide)	Way 1981; EPA 2008c
Steel manufacturing & heat-treating facilities, metal cleaning, electroplating		WHO 2004; Leduc 1984; EPA 2005
Ore-extraction (gold-mining, coke extraction)		WHO 2004; Leduc 1984; EPA 2005

Dyeing, printing of photographs		WHO 2004; EPA 2005
Production of resin monomers (acrylates)		WHO 2004
Pigments, paints	Ferrocyanides	Dzombak et al. 2006
Fire retardants		Little and Calfee 2002
Anti-caking agent for road salts		Dzombak et al. 2006
Detergents, dyeing of textiles		Dzombak et al. 2006
Pharmaceuticals (antibiotics, steroids, chemotherapy)		Dzombak et al. 2006
Fumigant/pesticide	Hydrogen cyanide, metallo-cyanide compounds	WHO 2004, Dzombak et al. 2006
Herbicides (dichlobenil, bromoxynil, bantrol)		EPA 2005, Dzombak et al. 2006
Road salts		EPA 2005
Production of other cyanides (e.g., sodium cyanide for gold mining)		EPA 2005, Dzombak et al. 2006
Pyrolysis of paper, wool, polyurethane		WHO 2004
Chelating agents for water and wastewater treatment		EPA 2005, Dzombak et al. 2006
Production of clear plastics		Dzombak et al. 2006
Methionine for animal food supplement		Dzombak et al. 2006
Wastewater Treatment Facilities (secondary treatment and/or disinfection w/ chlorine or UV)		Kavanaugh et al. 2003
Automobiles (with older or malfunctioning catalytic converters)		Voorhoeve et al. 1975; Karlsson 2004

With increasing uses and increasing production of cyanide we would expect that the amount of cyanide entering the environment would also be increasing (Way 1981). However, we have little data to ascertain if this is the case. According to the Toxics Release Inventory (TRI) data, total reported hydrogen cyanide releases have been increasing over the past 20 years (Figure 5). In 2008, some 4.5 billion pounds of HCN were released; over 432,000 pounds represent air emissions, while 58,000 pounds were discharged to surface waters (EPA 2008d). In comparison the long-term trend in releases to surface waters is declining, although this may not be a reflection of trends in actual ambient instream concentrations for several reasons. First, the data reflect one type of cyanide compound for which release data exists and does not include an assessment of the fate and transport of the released HCN including the ability of cyanide compounds to undergo transformation as under some environmental conditions that can increase or decrease its toxicological impact, and the TRI data does not include non-point sources of cyanide to the environment. Nonetheless, the TRI data, with its many caveats represents one of the only sources of data upon which trends in potential ambient cyanide can be discerned.

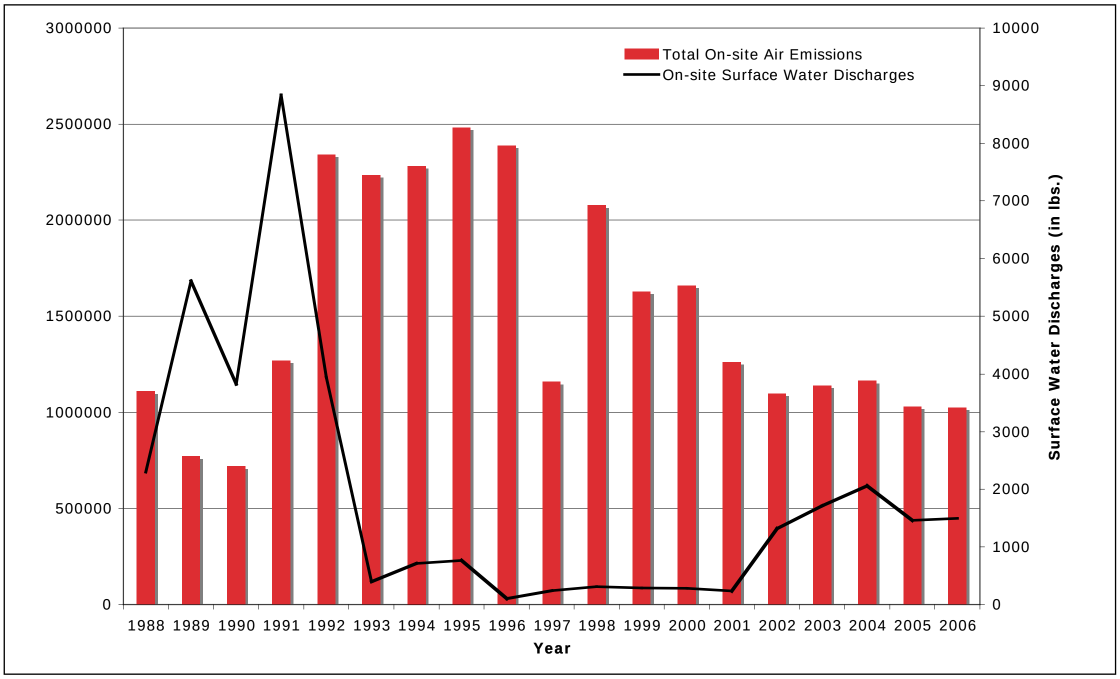

Figure 5. Toxics Release Inventory Data for HCN Releases in the United States to Air and Surface Waters, 1998 to 2006 (Source EPA 2008c).

The TRI data can also be used as indicator for understanding the geographic distribution of cyanide, in this case HCN, across the nation. The TRI data set, together with information on the distribution of manufacturers and user groups provide some insight into the distribution of cyanide sources and those areas where species might be at a higher risk of being exposed to ambient cyanide. While it is not clear that the volumes of cyanide discharged in these states typically resulted in aqueous concentrations that were problematic for listed species, the foregoing discussion illustrates that cyanide sources are widely distributed, and cyanide production and use is far from waning. On the contrary, cyanide production has increased in the past and is expected to increase in the future. As a result, we would expect listed aquatic resources are likely be exposed to one or more sources of cyanide during their lifetime. Due to the nature of the industrial sources, most exposure would occur in fresh water and marine coastal waters influenced by human activities. The predominant sources of cyanide to marine waters would be from direct discharges to marine waters (typically coastal outfalls), downstream transport from freshwater sources, and incidental releases from vessels (Dzombak 2006), which generally suggests that the further from shore a species or critical habitat occurs, the less likely it would be exposed to a wide variety of cyanide sources. However, with a large portion of cyanide entering the environment in gaseous form, we would expect some cyanide likely enters marine and fresh waters through atmospheric deposition.

Concentrations of Cyanide in U.S. Waters

As noted earlier, cyanide enters waterways through a variety of pathways and sources; however, the direct discharges (from point and nonpoint sources) pose the greatest concern for aquatic habitats because these sources are likely the dominant sources of cyanide loading to United States waters. To further characterize the exposure of listed resources in the aquatic environment, we asked whether and to what degree we would expect listed resources would be exposed to cyanide concentrations at or near EPA's recommended CCC or CMC for cyanide. We examined data in EPA's data base STORET (STORage and RETrieval data warehouse) for information on potential concentrations of cyanide in the environment, as well as individual studies of cyanide loading from various sources. Based on our evaluation, we expect listed resources will be exposed to a wide range of concentrations of cyanide, and a wide number of cyanide compounds with varying toxicity. We expect that most waters likely have some low-level background concentrations of cyanide at most times. When exacerbated by anthropogenic sources, in-water concentrations may exceed EPA's approved numeric criteria for cyanide and the averaging recommendations adopted in state standards.

Studies have detected low levels of cyanide as a natural condition in some waterways, likely resulting from plant and microbial input. There also appears to be a seasonal component to the cyanide loading in waterways, which presumably varies with cyanogenic plant production, atmospheric deposition and rainfall patterns. A study of the occurrence of cyanides (free and combined) in small streams in the North-West Germany, using a technique that allowed a detection limit of 0.1 µg/L, found annual values of total cyanide in rural watersheds was 3 µg/L, while mean annual values of total cyanide in industrial watersheds were 20 µg/L with values reaching over 200 µg/L (Krutz 1979 in Leduc 1981, Krutz 1981). Cyanide concentrations varied seasonally, with the lowest concentrations occurring in spring and late summer and highest concentrations occurred in winter. Krutz (1979 in Leduc 1981) calculated maximum winter loads at 6 g CN^-/day and summer loads at 0.2 g CN^-/day. Principal factors attributed to winter peak loading included increased potassium loads that induced cyanogenic microorganism activity and winter precipitation and runoff events that increased delivery of atmospheric cyanide and cyanide formed by plants and terrestrial microorganisms to the water. Seasonal peaks were more frequently observed in the small catchments, although seasonal peaks were also observed in medium to large sized catchments (Krutz 1979 and PPWB 1978 in Leduc 1981). On the other hand, Tarras-Walberg et al. (2001) found concentrations were highest when the river under study was in a low flow period. In many cases, the low flow period for a catchment would correspond with low-flows and peak vegetative growth within a basin. Consequently, small catchments tend to be more closely associated with streamside vegetation and allochthonous input of cyanogenic (and other) plants, which would explain the summer and low-flow peaks observed by Krutz (1981) and Tarras-Walberg et al. (2001).

Cyanide also enters waterways through the indirect pathway from airborne sources, such as burning waste biomass for energy conversion, crop burning, prescribed forest fires and wildfires, and through the atmospheric release of cyanide from industrial sources and the eventual transformation to aqueous cyanide. Barber et al. (2003) found that free cyanide concentrations in stormwater runoff collected after a wildfire in North Carolina averaged 49 µg/L, an order of magnitude higher than in samples from an adjacent unburned area (Barber et al. 2003). Atmospheric deposition of HCN may be one of the most significant sources of HCN to ocean waters, excluding coastal areas. However, according to Dzombak et al. (2006) the concentration of HCN in ocean waters is likely to be low (less than 1 µg/L than the criterion value for salt water).

Studies evaluating the direct discharge of cyanide to waterways indicate that the concentrations entering water are as variable as the sources themselves. Studies have shown that stormwater melting off roadside snow has a much greater capacity to accumulate and retain heavy metals and other pollutants than summer stormwater runoff. In a study of urban highway sites, concentrations of cyanide and metals were orders of magnitude higher than at the control sites and exceeded storm water (rain) runoff concentrations by one to two orders of magnitude. Cyanide concentrations, although demonstrating some variability, remained relatively constant at all sites (averaging 154 µg/L) or increased according increasing application rates of deicing salts that contained cyanide compounds as anti-caking agents (Glenn and Sansalone 2002). A study on the effect of cyanide on the anaerobic treatment of synthetic wastewater noted that cyanide is produced on an industrial scale of 2–3 million tons per year and, therefore is in many different industrial wastewaters. The concentrations encountered in industrial waste generally are in the range 0.01–10,000 mg/L, most of it in complexed forms of cyanide, which are less toxic than free cyanide but can transform to free cyanide or HCN. Cyanide contamination also occurs in the processing of agricultural crops containing high concentrations of this compound, such as cassava[12]. Systematic surveys of large wastewater effluents in Southern California suggest that free cyanide is routinely found in wastewaters, at low levels. In different years reported from 1992 – 2002, mean cyanide concentrations in effluents ranged from <2 to 30 µg/L (Steinberger and Stein 2003). Data from the US National Urban Runoff Program in 1982, revealed that 16% of urban runoff samples collected from four cities (Denver, Colorado; Long Island, New York; Austin Texas; and Bellevue, Washington) contained cyanide concentrations ranging from 2 to 33 µg/L (Cole et al. 1984 in ASTDR 2006). While demonstrating variability in the concentrations of cyanide found in some discharges, these studies also indicate that cyanide concentrations can be quite high at times.

The Difficulties of Measuring Cyanide in Water

Dzombak et al. (2006) refer to measuring cyanides as “a regulatory dilemma” because most analytical methods used in the field do not target specific cyanide compounds, rather the methods report various cyanide groups. EPA’s recommended aquatic life criteria are specified in terms of free cyanide, yet the conventional sampling methods provide for measurement of a group of cyanide compounds. Methods include total cyanide, weak-acid-dissociable cyanide (WAD), cyanide amenable to chlorination (CATC), available cyanide by ligand exchange, and free cyanide. Total cyanide, the most frequently conducted sampling method, measures free cyanide and metal-complexed forms of inorganic cyanide, while WAD measures weak metal-cyanide complexes plus free cyanide. Much of the older data available in such databases like STORET were measured and reported in terms of total cyanide, which although it could be used as a surrogate of the amount free cyanide in a sample, doing so would lead to an overestimate in the amount of free cyanide in the samples because total cyanide includes free cyanide, WAD cyanide plus the relatively non-toxic iron-cyanide complexes. When EPA published their recommended aquatic life criteria for cyanide in 1985, they recognized the incongruity between publishing numeric criteria for free cyanide, and the fact that no EPA approved sampling method was available at the time that would measure free cyanide (EPA 1985). Therefore, in 1985 EPA recommended that states apply the criteria to total cyanide, acknowledging that doing so may

[12] Although we are not aware whether the U.S. has any cassava processing plants, there are over 1,000 cyanogenic plants including many sorghum grains that may contribute to cyanide contamination when processed. Cassava is merely one of the best documented sources of cyanide contamination attributable to cyanogenic plant processing.

make the water quality standard over-protective. An approved method for measuring free cyanide is now available, but unfortunately a translator has not been developed to convert data on total cyanide to free cyanide (Kavanaugh et al. 2003).

At the same time, there is a concern over measurement precision with data found in sources such as STORET. Measurement precision varies among sampling methods and certain chemicals and procedures can interfere with measurements as well. Measurements are frequently conducted via colorimetric, titrimetric, or electrochemical finish techniques (Dzombak et al. 2006). Measurements of total cyanide are limited to detection in reagent water matrix of about 1 to 5 μg/L and do not measure: cyanates, thiocyanates, most organic-cyanide compounds, and most cobalt and platinum cyanide complexes (Dzombak et al. 2006). Problems with sample storage, regulatory criteria, and the methods for testing and their sensitivity are a concern (Eisler 1991; Dzombak et al. 2006). Eisler (1991) notes that due to the volatilization of cyanide, periodic monitoring is not informative (for example, monitoring once per quarter [for instance, see the permit requirements in EPA 2008e) except perhaps, where continuing or chronic conditions persist. Consequently, Eisler (1991) and others recommend that continuous monitoring systems are necessary, with particular emphasis on industrial dischargers, to understand the fate and transport, critical exposures, and relative contributions of human and natural sources of cyanide in the aquatic environment. The availability of data from case studies using continuous monitoring systems would significantly increase our understanding of cyanide in the aquatic environment, and provide us important exposure profiles for evaluating approved water quality standards. Unfortunately, we were not aware of any such data sets that we could examine as part of this analysis.

STORET – EPA's Main Repository for Water Quality Data

Since we do not have data on long-term studies using continuous monitoring systems to evaluate cyanide discharges, we conducted a query of EPA's STORET database to further characterize cyanide entering the action area for this consultation. STORET, EPA's main repository for water quality data, contains information on water quality collected from a variety of organizations across the United States, from small volunteer watershed groups to state and federal agencies (http://www.epa.gov/STORET/index.html). Our review of STORET data indicates that many dischargers reported no-detectable amount of cyanide in their samples, which in some case may have been a limitation of the sampling method and does not necessarily suggest that the water contained no cyanide or alternatively it may suggest that the discharges were free of cyanide – either way, we do not know. We searched the STORET database and found records spanning 1964 to 2008 (August), most of which were recorded as total cyanide. Some states of particular interest, like Washington, where NMFS has listed salmonids and where the TRI database suggests there have been large discharges of HCN to surface waters, were not represented in STORET. While data were available for several other states, data was often sparse for many of the coastal states where NMFS' listed resources occur. When we queried according to the data fields for "rivers, lakes, reservoirs, and canals" the database returned only one sample for Alaska and Oregon, 13 to 51 samples for states such as California, New Jersey and North Carolina. The largest number of samples in this category was from Florida. We compared the sample data to approved water quality standards for cyanide and found that 4 of 13 values reported for California (31%) exceeded the CCC and the CMC. Upon closer inspection it appears that most of the California data came from reservoirs and streams in the Mojave Desert that were presumably impacted by gold mining. The minimum value above the water quality standards was 300 µg/L and the highest reported value was 5,000 µg/L. For New Jersey, 17% of the reported values were above the CCC and the CMC. The highest reported concentration of cyanide above the approved water quality standard was 130,000 µg/L CN_T –there were two samples at this concentration in the data set, taken two weeks apart. Two months later, during the same year a concentration was sampled of 84,000 µg/L CN_T. All three of these samples were taken from the Ramapo River (near Mawhaw, New Jersey). Similarly, in Mississipi data show that state water quality standards were exceeded in 13% of the reported samples. When we queried STORET for data from marine waters we found only 5 reported values. All the samples were taken in Puerto Rico in about a 9-month span beginning late 2005. The mean concentration for this sampling station was 4.6 µg/L CN_T, four times higher than the approved water quality standard for marine waters, while the minimum reported concentration was 0 µg/L CN_T and the maximum concentration was 20 µg/L CN_T.

There are several very strong arguments that can be made questioning the utility of the STORET data for establishing an exposure profile for aquatic species. Not the least of which are the arguments that (a) the vast majority of information on cyanide in STORET is represented by total cyanide (CN_T) and a translator is yet to be developed that would allow us to determine the proportion of free cyanide (the most biologically toxic form) represented by the data on total cyanide, (b) the scarcity of data generally provides us little understanding of the spatial or temporal patterns of cyanide concentrations in United States waters, particularly since some states do not report their monitoring to STORET (or perhaps those states are not monitoring for cyanide), and (c) there are insufficient replicate data in STORET to provide any meaningful illustration of the trends in cyanide discharges within a particular locality. Despite the limitations of the data in STORET, it (with TRI data) represents some of the best available information we have on cyanide discharges across the United States. The STORET data however, does illustrate that listed resources may be exposed to a wide range of cyanide concentrations in receiving waters and that those concentrations may vary widely relative to EPA’s approved (and recommended) national numeric criteria.[13]

Given typical monitoring schemes in many permits the probability that a particular facility would detect an exceedence event is quite low. A typical permit may require sampling once a week, once a month, or less frequently, and will often conduct their sampling using grab samples[14] (see for instance, permit requirements in EPA’s 2008 Multisector General Permit). To determine whether or to what degree grab samples might detect events in which water quality criteria had been exceeded, we considered several scenarios. In the first scenario, we considered a facility that has 52 discharge events a year that result in elevated cyanide concentrations and assumed each discharge event lasted eight hours. In this scenario, there would be a 95% probability that the event would not be detected by a grab sample. Conversely, there would be a 5% probability that the event would be detected by a grab sample. If we increased the number of discharge events to 110 events per year with each event exceeding a particular criterion value and each

[13] An interesting question that merits exploration is whether EPA and/or the respective states considered many of the recorded peak discharges events that are included within the STORET database as violations of water quality standards.
[14] The grab sample technique is a rapid collection single point sampling method that does not integrate vertical or cross sectional variability, but captures point concentrations near the water’s surface.

event lasts 8 hours, there would be a 90% probability that the event would not be detected by grab samples, and a 10% probability that the event would be detected by a grab sample [15].

A discharge containing a high concentration of cyanide would have to occur for more than 180 days a year (24 hours/day) to have a high probability of detection, which suggests that random grab samples generally are not likely to detect an exceedence event. Therefore, the sample data we found in our query of STORET may not have been produced by truly random samples, but instead, were produced by samples taken after known discharges containing high concentrations of cyanide. The fact that some samples data points reported high concentrations of cyanide could also be attributed to serendipity during the timing of sampling, or it could be that the discharged concentrations are high for frequently long intervals of time (e.g., more likely than low concentrations).

Allowable averaging schemes contained in many NPDES permits would further mask the true distribution of sample concentrations to which listed resources are exposed. That is, recall that the approved standards include a provision that allows for the average of the 1-hour concentration for the CMC, and the average of the 4-day concentration for the CCC. A facility that takes ten samples a year may have one sample exceeding 200 µg/L CN and nine samples with "non-detectable" concentrations and that facility would still fall within the recommended limit recommend by EPA. Figure 6 illustrates three hypothetical scenarios that demonstrate how individual discharges may exceed the approved numeric standards for the cyanide CMC, but still fall within allowable standard when averaged accordingly. The three alternatives presented illustrate three scenarios, all with the same central tendency despite widely different sample distributions. The result is that all three scenarios would be presumed equal in perceived risk under the recommended averaging scheme, despite the actual and widely disparate concentrations to which fish and wildlife would be exposed. As a result of the averaging and infrequent sampling schemes, the power of the data to detect problems is exceedingly low, and the fact that so many samples reported in STORET are unusually high is cause for concern and suggests that in some areas cyanide concentrations may exceed the numeric values defined by the cyanide CCC and CMC fairly often. Consequently, based on the best available data it appears that at least some listed resources would be exposed to cyanide at concentrations well above the approved CMC of 22.36 µg/L CN and the CCC of 5.221 µg/L CN, and at a frequency and duration which may result in demonstrable harm to aquatic life.

Factors That Influence Cyanide Toxicity

The risk to aquatic environments from cyanide releases depends on several factors including: the cyanide compound and concentrations released, pH, presence of iron and other metallic trace elements, solar radiation, air and water temperature, and dissolved oxygen levels to name a few (Doudoroff 1976; Smith et al. 1978; Dzombak et al. 2006). There are several compounds in the cyanide group, with varying degrees of complexity. Cyanide is formed by carbon and nitrogen, attached by three molecular bonds ($C \equiv N$). Complex cyanide compounds are formed when one or more CN compound forms with other atoms, such as hydrogen ($H - C \equiv N$). The resulting

[15] The probabilities in this paragraph were derived using the equation: Probability = $1 - (1-p)^n$ where p is the probability of a water quality exceedance event in a sample, n is the number of samples, so $(1-p)^n$ is the probability of not detecting a water quality exceedance event, *and* then $1 - (1-p)^n$ is the probability of detecting a water quality exceedance event in a sample of size n. Adapted from McArdle 1990.

compound is HCN. Hydrogen cyanide (HCN(g)) is a gas that is miscible in water, and its water form, hydrocyanic acid (HCN(aq)), is weakly acidic and the most toxic cyanide compound. Other compounds include sodium cyanide (NaCN), potassium cyanide (KCN), adiponitrile ($C_6H_8N_2$), and copper cyanide (CuCN).

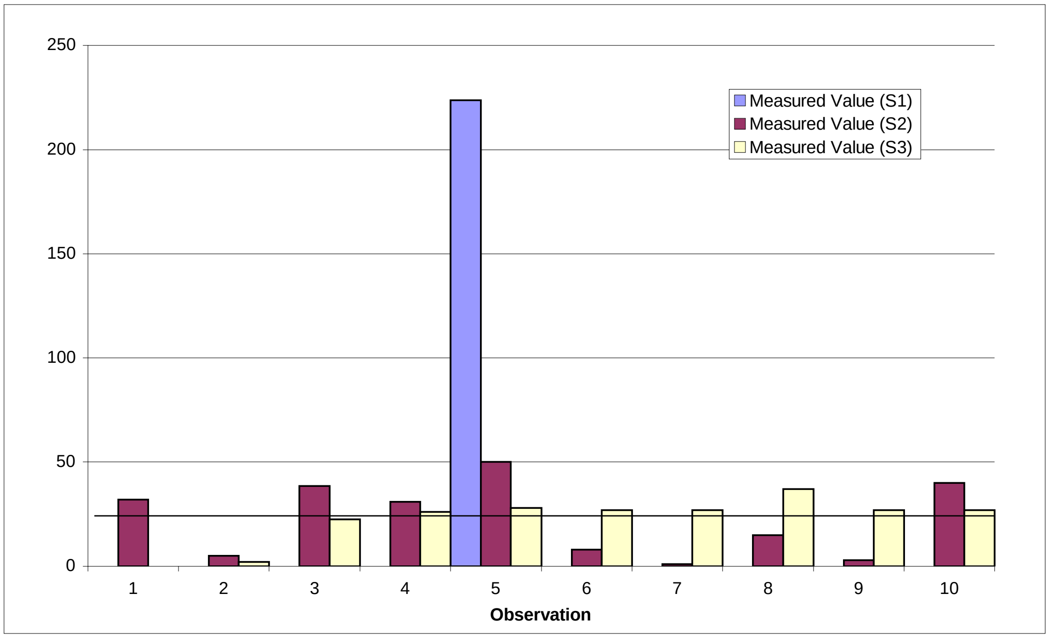

Figure 6. Three Hypothetical Discharge Scenarios that Comport with the Acute Water Quality Standard for Cyanide (Avg. CMC = 22.36 µg CN/L).

Free cyanide readily biodegrades, but the degradation is influenced by several factors including availability of oxygen, pH, carbon and nitrogen and the initial concentration of the cyanide compound released. Cyanide, through degradation, is converted to simple molecules like ammonia and carbon dioxide or it may it may be assimilated into the primary metabolism of bacteria, fungi, or plants (Dzombak 2006). Many forms of cyanide exist in the aquatic environment, including NaCN, KCN, metal-cyanide complexes, and organocyanides (e.g., acetonitrile), and metal-cyanide solids (e.g., ferric ferrocyanide). These forms have different chemical and toxicological properties. For instance, simple solids, like KCN and NaCN, are more soluble in solution and readily release free cyanide and HCN, which is the subject of this consultation. Solid forms of cyanide may exist in the soil of sites for years, and once exposed to water may result in dissolved cyanide reaching ground water and eventually surface waters (see Dzombak et al.'s [2006] discussion about the industrial legacy of cyanide box wastes at thousands of former manufactured gas plants in the United States). The chemical transformation of the cyanide compounds to HCN or CN^-, determines their toxicological significance to protected species, and their transport and fate in the environment.

The iron-cyanide complexes are the dominant form of cyanide found in soils and those most frequently encountered in dissolved form at concentrations in surface waters, making them the compound of most concern in managing water quality (Dzombak et al. 2006). The mobility of cyanide in soil and through groundwater depends upon precipitation, pH, the types of trace minerals present, and organic matter among other things. In water, cyanide transport, fate and toxicity vary according to volume dispersed, pH, temperature, mixing and turbulence, dissolved oxygen concentrations, form and abundance of alternative nitrogen sources, biological use, and incidental light as some cyanide complexes display photochemical reactivity (Leduc 1981; Kavanaugh et al. 2003; Dzombak et al. 2006). Regardless of what form of cyanide is introduced into a system, cyanide transformation mechanisms are variable according to environmental factors, and Kavanaugh et al. (2003) caution that managers need to acknowledge that multiple species of cyanide typically coexist, introconvert, and degrade in a system, and through its transformation the toxicological effect of cyanide may increase or decrease. Consequently, knowledge of cyanide compounds and their ability to undergo transformation is important to managing it in aquatic environments (Kavanaugh et al. 2003).

The Exposure Profile – Summarized

As noted earlier in this section (*Exposure – Cyanide Sources and Production*), we began our exposure assessment by examining general sources of cyanide across the United States, their spatial distribution and their production trends. We also examined available data to characterize CN concentrations in waters of the United States (the action area), and we compared the recommended (approved) numeric standards for cyanide to those values represented in data sets collected by EPA to determine if the numeric standards are representative of actual cyanide concentrations observed in United States waters. We also evaluated the ability of the data generally collected by EPA and authorized states to provide information that would help us make these comparisons.

Based on our analysis, we were unable to conclude that cyanide discharged in accordance with EPA's approved water quality standard has not co-occurred with listed and protected resources under NMFS' jurisdiction in the past, or that listed and protected resources would not be exposed to cyanide at some time in the future, such as over the course of the next 10 years[16]. As a result, we were unable to conclude that any particular listed or proposed resources should be excluded from our exposure analysis. The wide number of cyanide sources and uses and their broad geographic distribution suggests that some individuals of listed species, their designated critical habitat, and some individuals of species proposed for listing or their critical habitat proposed for listing, are all reasonably likely to be exposed to cyanide at some stage of their lives. Certainly, as the numbers of cyanide sources vary, the risk of exposure would also vary spatially and temporally across the action area. It appears that the potential for exposure may increase in urbanized areas, but rural areas are not free from potential sources of cyanide and some listed species would likely be exposed in these areas (e.g., gold mining and road maintenance activities are likely some of the sources in rural areas). In general anadromous fishes like salmonids and sturgeon, that traverse fresh and salt waters, would potentially be exposed to a greater number of cyanide sources throughout their life cycle, whereas listed marine species are more likely to be exposed to elevated concentrations of cyanide along the coasts than in deep or open ocean waters areas due to the combined effect of point and non-point sources from human activities. Both marine and fresh water species would likely be exposed to cyanide through deposition of

[16] We requested EPA identify a reasonable analysis period for their action, but did not receive assistance on this issue. Because the existing cyanide criteria were published in 1985 and have not been updated since, and we do not have any indication from EPA that they have plans to revisit their cyanide criteria, we used an analytical period of ten years. That is, our effects analysis considers the effects of continuing the approved cyanide criteria for an additional ten years into the future.

airborne releases. We did not find sufficient information to suggest that there were particular areas where listed species are not likely to be exposed to cyanide.

In a typical site specific assessment we would characterize the intensity of the listed resources and proposed resources exposure over time and space; however due to the inherent nature of this assessment, and the variability across sites and over time, such an estimate does not exist. Nor could we find data that we could use to assemble a case study of aquatic exposures in a particular space over a particular time. However, based on data collected by EPA (which is limited so the possibility for false positive errors (Type 1) and false negative errors (Type II) is high) it is clear that concentrations of cyanide have exceeded the approved standards in some locations and at some times. The data illustrate that the exceedances are sometimes orders of magnitude higher than the approved standards. Further, typical monitoring methods and the use of measures of central tendency on the collected data will often mask biologically important exposure scenarios. That is, we presented three alternative hypothetical sample data sets to illustrate that despite the distributions varied widely (i.e., even when individual events exceed the approved standards by 10 times) the perceived risk of the hypothetical sample sets would be presumed equal and in compliance with the approved water quality standard when using the central tendency as the measure of risk. In general, the monitoring and reporting practices routinely adopted in water quality standards severely reduce the utility of the data collected by EPA and states for characterizing typical exposure scenarios. Unfortunately, at the scale of this consultation and given the wide variability of the data available, it is not clear what might be a reasonable daily or longer-term potential dose for this analysis. Clearly, many factors influence the actual exposure of listed species in the wild and insufficient data are collected to evaluate the concentration, frequency and duration of allowable excursions, as well as the ambient concentrations to which authorized discharges are added. Simply, the criteria, as approved by EPA in state and tribal water quality standards, are the “protection level” to which the water quality based approach to pollution is applied. Absent better data to inform below and above criterion exposure events and other factors that influence exposure, we cannot confidently characterize the rarity or commonness of exposure scenarios that differ from the proposed criteria. Therefore, to anchor our response analysis for this consultation, we proceed with the core assumption that one or more life stages (all aquatic life stages) of all listed resources and resources proposed for listing would be exposed to cyanide at concentrations equivalent to EPA’s approved (and recommended) numeric water quality criteria. Since the CMC and the CCC represent the basis for administering water quality programs under the water quality-based approach to pollution control, including monitoring to determine whether waters are attaining designated uses, benchmarks for evaluating BMP performance in NPDES permits, evaluating whether waters should be listed as impaired, and as effluent limits for TMDL permits, we believe this is a reasonable core assumption for this analysis.

Response Analysis

As noted in our Approach to the Assessment, response analyses determine how listed resources are likely to respond after being exposed to an action’s effects on the environment or directly on listed species themselves. For the purposes of consultations on recommended or approved water quality standards, our assessments try to detect the probability of lethal responses, physiological responses, and behavioral responses that might result in reducing the fitness of listed individuals. Ideally, our response analyses consider and weigh evidence of adverse consequences, beneficial consequences, or the absence of such consequences.

It is important to begin these analyses by stating that, to the best of our knowledge, few data are available from the actual exposures of endangered or threatened species to cyanide in the laboratory or natural settings. We are aware of a few studies on rainbow trout, the resident form of *O. mykiss*; however, these studies are typically conducted on artificially propagated individuals that come from populations with a long history of artificial propagation such that their genetic make-up may be altered from their wild counterparts, and as a result there is some risk that their responses could differ from their wild counterparts. That said we have no information that would suggest this is the case and are assuming that there would be no difference in responses between artificially propagated individuals and wild individuals. Therefore, rainbow trout are the best surrogate available for predicting the response of wild steelhead, and many other species as well, because we lack species-specific data for several anadromous salmonids. We also have very little data for marine species as a group and no data on listed marine mammals. In fact, a recent reexamination of EPA's 1985 nationally recommended criteria for cyanide conducted by the Water Environment Research Foundation (Gensemer et al. 2007) concluded that "due to the lack of cyanide toxicity data for these species or reasonable surrogates", there was insufficient information available to evaluate the protectiveness of the saltwater cyanide criteria to threatened and endangered marine species. Instead, more research is needed on these species (Gensemer et al. 2007). Without empirical information on the actual responses of endangered and threatened species to cyanide, we reviewed the best scientific and commercial information available on the responses of fish and wildlife to cyanide. We also relied on estimates of sensitivity produced by EPA's Interspecies Correlation Estimations (ICE) model. We used this information to make inferences about the probable responses of endangered and threatened species when exposed to cyanide at the approved CCC and CMC.

Generalized Review of Responses

Individual aquatic organisms are exposed to cyanide by inhalation, ingestion, and absorption through epidural layers and mucus membranes. Cyanide is rapidly absorbed and distributed through the body. Once exposed, the primary manner of transport is via the bloodstream. In the bloodstream cyanide inhibits cellular respiration. Cyanide inhibits cytochrome c oxidase, an important hemeoprotein found in the mitochondria, by attaching to the iron in the protein it blocks the electron transfer to oxygen causing cellular respiration to cease. As a result many enzymes and biological systems are inhibited by cyanide, including succinic dehydrogenase, carbonic anhydrase and others (see Ballantyne 1987). Inhibition of cytochrome c oxidase activity, and the mitochondrial electron transport system will cause the cell to no longer aerobically produce ATP for energy, and the tissue then switches to anaerobic metabolism and the depletion of energy rich compounds (Eisler 1991; Dzombak et al. 2006). The result is rapid depression of central nervous system and the autonomic control of respiration. The heart is also a likely target of toxicity. Several species have shown consistently high concentrations within the myocardium, similar to brain concentrations, irrespective of the route exposure (Ballantyne 1987). Symptoms of acute poisoning in fish may include distress, increased ventilation – gill movement, surfacing, frantically swimming in circles at the surface, violently swimming against the bottom, convulsions, tremors, and finally death (Leduc 1981).

As a powerful and rapid asphyxiant, cyanide will cause death in a manner of minutes by hypoxic apoxia at lethal concentrations. Releases of cyanide at extreme lethal doses are likely rare based on known fish kills and STORET data, but they do occasionally occur. However, when they do occur, massive fish kills result. Some such events occurred in:

- Wissahickon Creek, Pennsylvania, where more than 1,000 fish were killed in 2006 due to the dumping of about 25 gallons of potassium thiocyanate, which is suspected of having interacted with chlorine in the nearby wastewater treatment plant (EPA 2006).
- Alamosa River, Colorado, where the Summitville Mine was responsible for contaminating 17 miles of the river and killing more than 15,000 trout in 1990 due to the escape of cyanide-laden pit waters. By the 1992, the site was abandoned by the mining company and was a notable superfund site, at high risk of additional leaks (Gavin 2004).
- Fall River, Oregon, where more than 22,000 trout died in 2002 when 1,000 to 2,000 gallons of fire retardant, which was released during fire fighting activities reached the waterway. The fire retardant was mixed with sodium ferrocyanide, which was used as corrosion inhibitor to protect the tanks the retardant was stored in (ODFW 2002).

Other events like these have occurred in the United States, and there have also been several events in other countries such as Ghana, Kyrgyzstan, and Canada, to name a few. Such events, while severe when they occur, tend to occur infrequently. Typically, we would find cyanide at much lower concentrations in the environment.

Cyanide although a potent asphyxiant, is also rapidly detoxified. The major determinant of the severity and rapidity of a response depends upon the rate of absorption versus the rate of detoxification, which are influenced by the rate and severity of exposure. Detoxification occurs primarily through the enzymatic transformation to thiocyanate, which is excreted by the kidney (Ballantyne 1987).

At sublethal doses, individuals may act stunned, which is why cyanide is widely used for the collection of tropical fish for aquariums. Sublethal doses can also inhibit reproduction, metabolic rate, egg production, spermatogenesis, oocyte development, lead to tissue necrosis, aggressiveness, impair food capture, and interrupt ion regulation and swimming ability (see Doudoroff 1976, Kimball et al. 1978, Leduc 1984, Eisler 1991). On the other hand, low-level exposure may also stimulate growth (Negliski 1973 in Dzombak et al. 2006; McCracken and Leduc 1980). Whether there are concomitant adverse effects to other physiological development process associated with growth stimulated by cyanide exposure is unclear. Rapid detoxification occurs at lower doses, as cyanide is metabolized to thiocyanate by two enzymes that are widely distributed in the body, and then excreted in urine over a period of days. Although thiocyanate (SCN^-) is the principle form of cyanide that is eliminated, it can also accumulate in tissues and is known to have antithyroidal properties. SCN^- inhibits iodine uptake by thyroid tissues and disrupts thyroid hormone homeostasis, which can result in the development of goiter. Cyanide does not bioaccumulate through the food web; however, the damage associated with prolonged exposure at low levels, recovery, and re-exposure may be cumulative. There is no evidence to suggest cyanide is mutagenic or carcinogenic (Ballantyne 1987).

Calculating a Response

Studies on the responses of listed resources and resources proposed for listing to cyanide and cyanide compounds are few. Directed studies on listed and proposed resources would generally rank highest for consideration, provided the studies were carefully designed, had large sample sizes (with small variances), and measured cyanide using a reliable test method. Such studies would generally provide the most reliable indicator of a listed species response, when exposed to cyanide in the wild. However, because data are not available for large number of fish and wildlife species EPA's *Guidelines* establish some minimum standards for deriving water quality criteria.

Generally, EPA would use the GMAV, which are calculated as the geometric means of the available SMAV to set the acute criterion, although this was not the case for their recommended aquatic life criterion for cyanide in fresh water. EPA calculated the acute freshwater value or CMC for cyanide (22.36 µg/L) to protect the recreationally and commercially important rainbow trout, the most sensitive of the species tested. Data were available for 15 different genera, and the most sensitive species of those tested was rainbow trout. At the time, rainbow trout was classified as *Salmo gairdneri*, and the other species in the same genera for which EPA had test data was the Atlantic salmon, which incidentally had a SMAV double that rainbow trout. Therefore, EPA chose to use the rainbow trout SMAV to set the acute criterion for cyanide. The acute criterion for saltwater was calculated using the GMAV from eight different genera, with *Cancer irratus* representing the lowest ranked GMAV. EPA then divided the FAV by 2[17] to derive the CMC. There was however, insufficient chronic toxicity data available to meet the minimum standards established by the *Guidelines* therefore EPA applied the ACR to the FAV to estimate the final chronic value. Unless there are other data to suggest the FCV is not sufficiently protective, the CCC is set to the FCV. For cyanide, once the ACR for four species was calculated, EPA took the geometric mean of the four freshwater species to derive the final ACR. Next the FAV was divided by the final ACR, to derive the final CCC. For saltwater, the CCC was set equal to the CMC because it was assumed that the acute sensitivity of the rock crab was a better indicator of the chronic sensitivity of the species than would be obtained otherwise. Table 36 contains a summary of the cyanide water quality standards and the top-ranked values used to calculate the CMC and the CCC.

Table 39. Summary of cyanide test results and subsequent water quality criteria[1].

GMAV	Fresh water		Saltwater	
Rank	**Genus**	**GMAV (µg CN/L)**	**Genus**	**GMAV (µg CN/L)**
4	Lepomis	99.28	Mysidopsis	118.4
3	Perca	92.64	Menidia	59
2	Salvelinus	85.80	Acartia	30
1	Salmo	63.45	Cancer	4.893
FAV (calculated from GMAVs)		62.68		2.030
FAV (SMAV for rainbow trout)		44.73		
CMC		22.36		1.015
ACR		8.568		2

[17] By dividing the FAV by 2, EPA believes that they have derived a CMC concentration "that will not severely adversely affect too many of the organisms (EPA 1985)".

CCC	5.221	1.015

[1]Table adapted from Gensemer et al. 2006; data from EPA 1985.

Acute Toxicity

Knowledge of the acute lethal effects of cyanide on fish has been gained through observations following accidental spills, intentional field application for lake/stream management and controlled laboratory studies. Cyanide is highly toxic with a relatively short half-life. At high levels of exposure, acute toxicity occurs rapidly (Leduc 1984). During intentional field applications exposed fish were observed exhibiting symptoms that include increased ventilation, surfacing, gulping for air, frantic swimming in circles, convulsions and tremors prior to death (Leduc 1984). Laboratory tests under controlled situations have revealed that not all life stages of fish are equally sensitive to acute cyanide exposure, that cyanide toxicity can be modulated by abiotic factors, and that there is a wide range in sensitivity among aquatic organism. For instance, Smith et al. (1978) demonstrated that bluegill, yellow perch, and brook trout juveniles were more sensitive than newly-hatched fry, where, as swim-up fry were the most sensitive fathead minnow life stages.

EPA and the Services conducted an extensive search for data for the consultation, which included a review of studies that had been used in the derivation of the cyanide criteria in 1985 and any new studies that had been conducted since 1984. EPA compiled toxicity data for 83 species of aquatic animals and plants (61 freshwater species and 22 saltwater species) as part of their BE for the cyanide consultation (EPA 2007). Based on this compilation, there appears to be a large range in sensitivity between the most sensitive (rock crab LC_{50} 4.89 µg CN/L) and the least sensitive species tested (river snail LC_{50} 760,000 µg CN/L). Freshwater species represented 9 phyla, 15 classes, 29 orders, 36 families, and 52 genera. Fishes were among the most sensitive freshwater taxa although there was substantial variability in sensitivity. Among the 24 freshwater fish species included in the list there was a 33 fold difference in sensitivity between the most sensitive (rainbow trout, *Oncorhynchus mykiss*, LC_{50} 59 µg CN/L) and the least sensitive (bata, *Labeo bata*, LC_{50} 1970 µg CN/L). The 8 most sensitive fish species belong to 3 different families, Salmonidae (3 species, 3 genera), Percidae (2 species, 1 genera), and Centrarchidae (3 species, 3 genera). Because of the relatively low number of species that have been tested within these families it is difficult to get a sense of the amount of intra-family variability in species sensitivity on the low end of the species sensitivity distribution. By contrast, the family cyprinidae was well represented with 10 different species representing 8 genera. Among those 10 species there is an 18-fold difference in sensitivity between the most sensitive (roach LC_{50} 108 µg CN/L) and the least sensitive (bata, *Labeo bata*, LC_{50} 1970 µg CN/L) species. Because of pronounced intra-family variation it is unlikely that the 8 species within the 3 most sensitive families represent the most sensitive species within those families.

Within the compiled data set, empirical data on the acute effects of cyanide was available for only two biological species under NMFS' jurisdiction—steelhead (representing 11 listed species (DPSs) of *O. mykiss*) and Atlantic salmon (*Salmo salar*)[18]. Consequently, EPA estimated EC_As

[18] Atlantic salmon are jointly managed with FWS. This species is addressed in the FWS' biological opinion on this action.

were calculated using ICE or SSD for six biological fish species under NMFS' jurisdiction, representing 19 listed species (ESUs and DPSs).

Steelhead/Rainbow Trout. Previous work by EPA and others suggest that rainbow trout are the most sensitive freshwater test species to cyanide. That is, the concentration of cyanide that induces mortality is lower than it is for many other species, with warmwater species generally exhibited greater tolerance. We found one additional study on the acute response of rainbow trout to cyanide that has been conducted since EPA calculated the 1984 aquatic life criteria. The study by McGeachy and Leduc was published in 1988 and analyzed the influence of season and exercise on the acute responses of rainbow trout to cyanide. The other studies on the lethal responses of rainbow trout to cyanide were available at the time EPA published their cyanide criteria in 1985. In 1985, EPA chose to use only 4 values for calculating the SMAV for rainbow trout (Table 37). EPA's reasoning for choosing those studies at the time, was because in a comparison of acute toxicity values for fishes they confirmed what Doudoroff (1976) had concluded earlier, that static toxicity tests generally produced higher response values than flow-through tests of equal, fairly prolonged duration (EPA 1985). As a result, they based the SMAV on the results of flow-through tests in which the concentrations were measured (EPA 1985). This comports with direction provided by the *Guidelines* (Stephan et al. 1985) which states:

- For some highly volatile, hydrolyzable, or degradable materials it is probably appropriate to use only results of flow-through tests in which the concentrations of test material in the test solutions were measured often enough using acceptable analytical methods
- For each species for which at least one acute value is available the SMAV should be calculated as the geometric mean of the results of all flow-through tests in which the concentrations of test material were measured.

Thus, the estimated mean acute value influences the estimated assessment effects concentration and the preliminary screen for making Section 7 effects determinations (also the estimated level of protection) under the *Method Manual*. For instance, Table 38 compares acute data from: all referenced studies used by EPA in their 1985 published recommendation for cyanide and used by Gensemer et al. (2007) in their recent review of the cyanide criteria, an approximation of EPA's calculated LC_{50} that they used in the BE to make their effects determination[19], and two alternative data sets to calculate the SMAV and EC_As for steelhead. Using only flow-through test data EPA (1985) and Gensemer et al. (2007) derived SMAVs of 44.73 µg CN/L and 46.53 µg CN/L, respectively. The difference in SMAVs is attributed to Gensemer et al.'s (2007) addition of values from the flow-through tests conducted by McGeachy and Leduc (1988), which were not available at the time the criteria document was published. Because the precise values EPA (2007) used in their BE calculation were not clear to NMFS when there were multiple test values available within a particular study, we used data values that allowed us to approximate their final LC_{50} value. For instance, we are aware EPA used data from Markings et al. (1984) but are not clear what particular values influenced their final LC_{50} calculation.

Marking et al. (1984), Bills et al. (1977), and Skibba were not used in the calculation by EPA (1985) or Gensemer et al. (2007) because the data were derived from static tests, which as noted

[19] EPA calculated a mean LC50 for steelhead (rainbow trout) of 59.22 (µg CN/L). We tried, but could not precisely replicate this value.

earlier tend to produce responses at higher concentrations. Neither EPA (1985) nor Gensemer et al. (2007) stated why the data from Dixon and Sprague (1981) were not used in the calculation. Although these studies were not used in the mean LC_{50} calculation, EPA (1985) and Gensemer et al. (2007) considered the studies as “other data”.

Alternatives 1 and 2 in Table 38, NMFS followed suit with the *Guidelines* and relegated static-test data for later consideration but did not include these data in the LC_{50} calculation. The primary difference between Alternatives 1 and 2, however, was in the test data we included from McGeachy and Leduc (1988). McGeachy and Leduc (1988) compared the toxicity of cyanide under different swimming conditions-- “exercised” versus “non-exercised” conditions. The non-exercised trout were placed in white polyethylene tanks and surrounded with Styrofoam and black plastic to minimize disturbance. It appears that Gensemer et al. (2007) chose to use the data from “non-exercised” fish in their calculation. For comparison, we used only the data from “exercised” trout in Alternative 1 because these fish were kept in more realistic test conditions (i.e., more natural), whereas all the data from McGeachy and Leduc (1988) are used in Alternative 2.

Table 40. Comparison of Toxicity Values To Support Species Mean Acute Value Calculations for Rainbow Trout

Mean LC_{50} Value	**LC_{50} Value used to calculate SMAV (µg CN/L)**					**Reference**
	EPA 1985	**Gensemer et al. 2007**	**EPA 2007***	**Alt. 1**	**Alt. 2**	
90			90			Bills et al. 1977
57	57	57	57	57	57	Smith et al. 1978; Broderius and Smith 1979
27	27	27	27	27	27	Kovacs 1979
40	40	40	40	40	40	
65	65	65	65	65	65	
98			98			Dixon and Sprague 1981
98			98			
97						
96						
97						
67						
83						
95						
46			46			Marking et al. 1984
52			52			
54			54			
62			62			
75			75			
55			55	55	55	McGeachy and Leduc 1988
53		53	53		53	
50				50	50	
42		42	42		42	
56				56	56	
53		53	53		53	
56				56	56	
66					66	
97			97			Skibba 1981

64.28	44.73	46.53	59.15	49.28	50.49	SMAV
28.32	19.70	20.50	26.06	21.71	22.24	EC_A using EPA's 2.27 LTAF
49.07	34.15	35.52	45.15	37.62	38.54	EC_A using Gensemer et al. LTAF of 1.31
0.79	1.13	1.09	0.86	1.03	1.01	R

*Values used in the calculation were not provided, and are assumed approximately equivalent to those provided herein.

As noted earlier, this risk paradigm was designed to estimate the relative risk of a chemical, such that the farther away from 1 an R-value, the greater the assurance the assessor would have in their Section 7 effect determination. However, the strength of the EC_A (and the effects determination) depends on the availability of pertinent evidence, and ultimately on the identification, appraisal, assimilation, and interpretation of that evidence. A strict interpretation of the risk paradigm indicates that four of the six scenarios illustrated in Table 37 would warrant a preliminary "likely to adversely affect" determination until additional data is provided that demonstrates otherwise (e.g., "other data" not used in the SMAV calculation, and a closer review of the data used in the LC_{50} calculation). While the risk ratio is merely an indication of potential risk, it is clear that the values chosen to calculate the species' LC_{50} value can influence the preliminary screen risk prediction. Based on our comparison, it also appears that the values EPA used to calculate the SMAV for rainbow trout was conservative, given the larger data set.

Nevertheless, we are concerned that this analytical approach can generate misleading results by ignoring meaningful differences among studies. That is, when the data are normalized first by calculating the geometric mean of the LC_{50}s without regard to the underlying distribution of the data, resolution is lost. In addition to examining the pooled data set to see that it is comprehensive, we must also closely examine the distribution of the underlying data, and differences in test methods (doses, schedules, modes of treatment, etc.) to ensure important differences in data are not drowned in a single estimate generated from a pooled data set (Lau et al. 1998). Uncertainty is incorporated in our analysis when we "focus on the tails of the distribution rather than on the measure of central tendency (the mean or best estimate)…. (Taylor and Wade 2002)." A careful examination of the pooled data set is warranted to ensure we have appropriately incorporated uncertainty and to ensure that the method provides a high degree of conservatism (e.g., errs on the side of the protecting the species when data are not sufficient to reasonably conclude the action is "not likely to adversely affect" the listed species or its critical habitat). When we examine the distribution of the data for rainbow trout we see that the lowest test value presented by Kovacs (1979) approaches the CMC (LC_{50} at 6 ºC = 28 µg CN/L). When we apply EPA's extrapolation factor of 2.27 to the lowest LC_{50} value available for rainbow trout we can estimate of the lowest concentration likely to be lethal to 0 to 10 percent of the population. The resulting LC_{10} for very cold-water conditions (6 ºC) is 12 µg CN/L. That is, when exposed to as little as 12 µg CN/L in cold waters, as much as 10% of the exposed threatened and endangered steelhead may die.

EPA derived the lethality threshold adjustment factor, 2.27, from a combined data set on fresh water and marine fish and invertebrates, a number of chemicals tested, as well as whole effluent data. In comparison, Dwyer et al. (2005) looked at five chemicals and seventeen species, including a few listed species, and also found the average factor to calculate a no- or low-effect concentration varied among pollutants and species (0.50 - 0.66), with the geometric mean for all species as 0.56 (f^{-1} = 1.8). More recently, DeForest et al. (in Gensemer et al. 2007) compiled concentration-response curves for rainbow trout, using data from McGeachy and Leduc (1988), and Kovacs and Leduc (1982), estimated the lethality threshold adjustment factor as 0.76 (f^{-1} = 1.316). Applying the extrapolation factor from Dwyer et al. (2005) results in a low effect concentration of about 16 µg CN/L, and DeForest et al. (in Gensemer et al. 2007) would result in a low effect concentration of 21 µg CN/L. DeForest et al. (in Gensemer et al. 2007) estimated the mean LC_{01}:LC_{50} ratio based on the steepness of the concentration-response curves to produce an estimated effect level lower than the LC_{01} (DeForest et al. in preparation, cited in Gensemer et al. 2007). Using DeForest et al.'s calculated adjustment factor, we would expect that 1% of the sample population would be expected to die as a result of their exposure at that calculated cyanide concentration.

In a separate analysis of the lethality threshold adjustment factor, FWS found EPA's 1978 data set upon which the 2.27 was derived from widely varied data and thus recalculated the adjustment factor as a standardized LC_{50}/LC_{10} using 62 acute exposure-response regression equations for cyanide (Appendix C). FWS' recalculated adjustment factor calculated for rainbow trout was 1.14. Had EPA used this, or any of these revised adjustment factors, more species would have been screened out as not likely to be adversely affected by their exposure to cyanide at the CMC. This further suggests that at least for cyanide, EPA's lethality threshold adjustment factor of 2.27, despite having introduced an additional source of uncertainty into estimates of the EC_A, likely produced preliminary estimates in accordance with the approach in the *Methods Manual* that erred on the side of inclusion rather than screening out species. Again, if we look at the distribution of the acute data for rainbow trout, using Kovacs' (1979) LC_{50} of 28 µg CN/L, which was derived in very cold water temperatures, and apply EPA's adjustment factor of 2.27 then an estimated 1 to 10% of individual steelhead may die when exposed when exposed to as little as 12 µg CN/L in very cold waters (6 ºC or less). Alternatively, if we apply the FWS' recalculated adjustment factor for cyanide to the same data, then the LC_{10} concentration would be above the CMC (at 24.56 µg CN/L).

Other Pacific Salmon Species. Based on species-specific estimates for coho and Chinook salmon and estimates for the genus *Oncorhynchus*, ICE predicts that coho, Chinook, sockeye, and chum salmon are relatively more sensitive than steelhead to cyanide (see Table 38). That is, based on the lower bound of the ICE predicted LC_{50}, coho, Chinook, sockeye, and chum salmon are all "likely to be adversely affected" when exposed to cyanide. Of these four fish within the genus *Oncorhynchus*, EPA's ICE results suggest that coho salmon are the most sensitive Pacific salmon with a predicted acute EC_A of 15.51 µg CN/L, with an estimated LC_{50} of 53.16 µg CN/L. In comparison, also using ICE, DeForest (pers. comm.) estimated the LC_{50} for coho salmon as 41.9 µg CN/L and the LC_{50} for Chinook salmon as 50.9 µg CN/L (Table 5). When we recalculated the EC_A using the divisor 2.27 and the expected LC_{50} value, for both species, the EC_A fell above the CMC using EPA's LC_{50} value, but below the CMC using the estimated LC_{50} calculated by DeForest (pers. comm.). Whereas, DeForest concluded, based use of the LC01 divisor, 1.316, that coho salmon and Chinook salmon were protected by the CMC (both LC01 values are greater than 30 µg CN/L). Based on the work by Gensemer et al. (2007), and DeForest (in Gensemer et al. 2007), the ICE model is likely to conservatively overestimate the sensitivity of most species (i.e., produce lower LC_{50} values than would likely be measured). DeForest (pers. comm.) concluded, based on his analysis of the empirical cyanide SMAVs, that there is an eight percent probability that a fish species would be more sensitive to cyanide than rainbow trout; whereas, if the ICE-estimated LC_{50} values are considered in the SSD, then there is about a 20% probability that a fish species would be more sensitive than rainbow trout. Based on our recalculations and information from DeForest (pers. comm.), and EPA's use of the lower 95 confidence level to calculate the EC_A for these species, it appears that EPA's preliminary effects determination that these species should not be screened out would be conservative (i.e., that is it erred on the side of protecting listed species given the uncertainty in the estimates).

The Influence of Other Data

The preliminary screen in the *Methods Manual* was designed to be a first step for reviewing robust response data, and conclusions based on this screen should be carefully reviewed by rechecking each step. That is, studies that have been dismissed because they do not meet basic requirements for the calculation of EC_A require review as "other data". EPA's *Guidelines* explicitly state that

> *Pertinent information that could not be used in earlier sections might be available concerning adverse effects on aquatic organisms and their uses. The most important of these are data on cumulative and delayed toxicity, flavor impairment, reduction in survival, growth, or reproduction, or any other adverse effect that has been shown to be biologically important. Especially important are data for species for which no other data are available. Data from behavioral, biochemical, physiological, microcosm, and field studies might also be available. Data might be available from tests conducted in unusual dilution water, from chronic tests in which the concentrations were not measured, from tests with previously exposed organism, and from tests on formulated mixtures or emulsifiable concentrates. Such data might affect a criterion if the data were obtained with an important species, the test concentrations were measured, and the endpoint was biologically important (Stephan et al. 1985).*

According to the *Guidelines*, EPA ought to consider "other data" in its decision to recommend a criterion. Unfortunately, it's not apparent that this "other data" influenced EPA's final value for the cyanide CMC (or CCC) in their 1985 cyanide recommendation. Nor is there evidence to suggest that particular states incorporated the "other data" in their final state water quality criteria, such that particular exceptions or special management actions were written into the final adopted water quality standard, when applicable. We were particularly interested in the effects of temperature and dissolved oxygen on EPA's decision to recommend the cyanide criteria because these are two factors known to affect cyanide toxicity, and because studies that have directly explored these relationships with listed resources (steelhead and Atlantic salmon). We explore this "other data" in the following sections.

1 Table 41. Species Specific Toxicity Estimates (EPA 2007**).

Species*	Saltwater v. Freshwater Exposure	Acute EC_A (µg CN/L)	Chronic EC_A (µg CN/L)	Species Specific Toxicity Data	Estimation Method Used	Taxon Represented by EC_A
Coho salmon	SW (adult & smolt) FW (all life stages)	15.51	3.33	N	ICE	*Oncorhynchus kisutch*
Chinook salmon	SW (adult & smolt) FW (all life stages)	16.26	3.49	N	ICE	*Oncorhynchus tschawytscha*
Chum salmon	SW (adult & smolt) FW (all life stages)	21.41	4.60	N	ICE	*Oncorhynchus* (genus)
Sockeye salmon	SW (adult & smolt) FW (all life stages)	21.41	4.60	N	ICE	*Oncorhynchus* (genus)
Steelhead	SW (adult & smolt) FW (all life stages)	26.08	9.80	Y		
Shortnose sturgeon	SW (adult & juveniles) FW (all life stages)	29.28	6.39	N	SSD	*Actinopterygii* (class)
Green sturgeon	SW (adult & juveniles) FW (all life stages)	29.28	6.39	N	SSD	*Actinopterygii* (class)

2 *EPA also included Totoaba. This is a foreign-listed species and would not be adversely affected by the proposed action because it is not exposed to United States waters.
3 **Data are from EPA 2007. Life stages exposed were adjusted to account for all possible life stages that could be exposed in fresh water and salt water.

Table 42. Comparisons of LC50 values for coho and Chinook salmon (µg CN/L)

Species	Estimated Mean LC_{50}	Lower 95% CL	Estimated EC_A using expected LC_{50}	Estimated LC01 (f^{-1}= 1.316)
Coho salmon	53.16*	35.21*	23.42*	40.40
	41.9**	25.2**	18.46	31.84**
Chinook salmon	64.35*	36.91*	28.35*	48.90
	50.9**	24.7**	22.42	38.68**

**Data from DeForest, pers. comm.
*Data from EPA 2007

The Influence of Temperature on Tolerance Limits

As a general matter the tolerance of fish to many pollutants tends to decrease with increases in water temperatures. Studies have demonstrated that the effects of temperature on the toxicity of cyanide can vary with concentration and temperature such that cyanide toxicity increases at high temperatures and at very low temperatures. Studies that have evaluated the effects of cyanide at high temperatures have found that the toxic action of cyanide increases with increasing temperatures, but many of these studies were conducted with extremely high doses of cyanide (see Doudoroff 1976). Early studies indicated that the 72-hour median lethal concentration or tolerance limit increased almost threefold with increased temperatures, when rainbow trout were exposed to test temperatures ranging from 4 to 20 ˚C (Great Britain, Ministry of Technology 1969 in Doudoroff 1976). Unfortunately, it is not clear what cyanide concentrations were used in the Great Britain study (Doudoroff 1976). Later, Kovacs (1979) confirmed that there are significant differences in 96-hour LC_{50} values between 6, 12 and 18 ˚C, such that it took 2.4 times less cyanide to kill 50% of the trout in 96 hours at 6 ˚C than it did at 18 ˚C. One of the primary differences between work by Kovacs (1979) and earlier researchers is the rate and concentration of the doses administered. Kovacs (1979) administered cyanide at slowly lethal concentrations, whereas earlier studies tended to focus on rapidly lethal concentrations, suggesting that the potency of cyanide is both temperature and concentration dependent. Doudoroff (1976) suggested that the lethal response at low temperatures is likely a result of a decrease in the rate of detoxification at lower temperatures, which is affected by the decline in the metabolic rate at lower temperatures. Death at lower temperatures may also be caused by the disruption of cytochrome oxidase activity (Kovacs 1979).

Since steelhead rearing and spawning typically occurs in temperatures ranging from about 4˚C to about 15˚C (Barnhart 1991). Consequently, increased cyanide toxicity at lower temperatures could have serious consequences for steelhead fitness. We chose four river basins that we felt were representative steelhead rivers—one from each of the western states, Oregon, Washington, Idaho, and California, where there are listed steelhead populations—and examined the mean monthly water temperatures for comparison to the low temperatures measured by Kovacs (1979). Figure 7 compares the mean monthly water temperatures to the generalized life history stages of steelhead in the Clearwater River, Idaho. Steelhead in this system compose two-runs, an “A” and “B” run, which are distinguished according to their size and ocean life history. Spawning occurs from mid-April to late June, with “A-run” fish returning after one year in the ocean and “B-run” fish returning after two years in the ocean. Due to the long freshwater rearing period of juvenile steelhead and the long holding period of adults, at least two to three age classes of steelhead can be found in the basin during winter. As illustrated in Figure 7, winter water temperatures are at or below 6˚C for several months each year (about 5 months). Similarly, water temperatures are at or below 6˚C in the Puyallup River in Washington for about three months when adult and juvenile life stages would be in the basin (Figure 8). In the North Umpqua River in Oregon water temperatures are at or below 6˚C for about four months of the year, when additional life stages are present including migrating and spawning adults, eggs, fry, and juvenile fish (Figure 9). In the Klamath River in Oregon/California average water temperatures are below 6˚C for a brief period of time (about a month), but these temperatures occur when adults are migrating and spawning, and juvenile steelhead are rearing (Figure 10). Due to the iteroparous life history of steelhead and the propensity for multiple juvenile age-classes to rear together, these basins would generally have at least two age-classes but may have four or more age-classes in the basin during winter.

We looked but did not find information to suggest that states or EPA would generally modify the cyanide water quality standards to minimize the impacts to salmonids in cold water. We looked for this information particularly in state water quality standards for Idaho, California, Washington and Oregon. Generally, we found that when states modified EPA's nationally recommended criteria they did so to increase the cyanide concentration, not decrease acceptable limits. However, we did not search specific permit conditions to evaluate whether permits were adjusted to account for increased toxicity of cyanide during low temperatures.

All of the Pacific salmonids under NMFS' jurisdiction, green and shortnose sturgeon, and Atlantic salmon are exposed to very cold water temperatures during their life cycle. We would not expect that the general response of increased toxicity at low temperatures is species specific response, but is a generalized physiological response of fish that occupy cold streams. The low acute response of steelhead is likely a reasonable predictor of other Pacific salmonids, but we do not know the lowest response value of sturgeon or Atlantic salmon nor do we have a suitable surrogate to estimate this response. Clearly, more studies are warranted in this area.

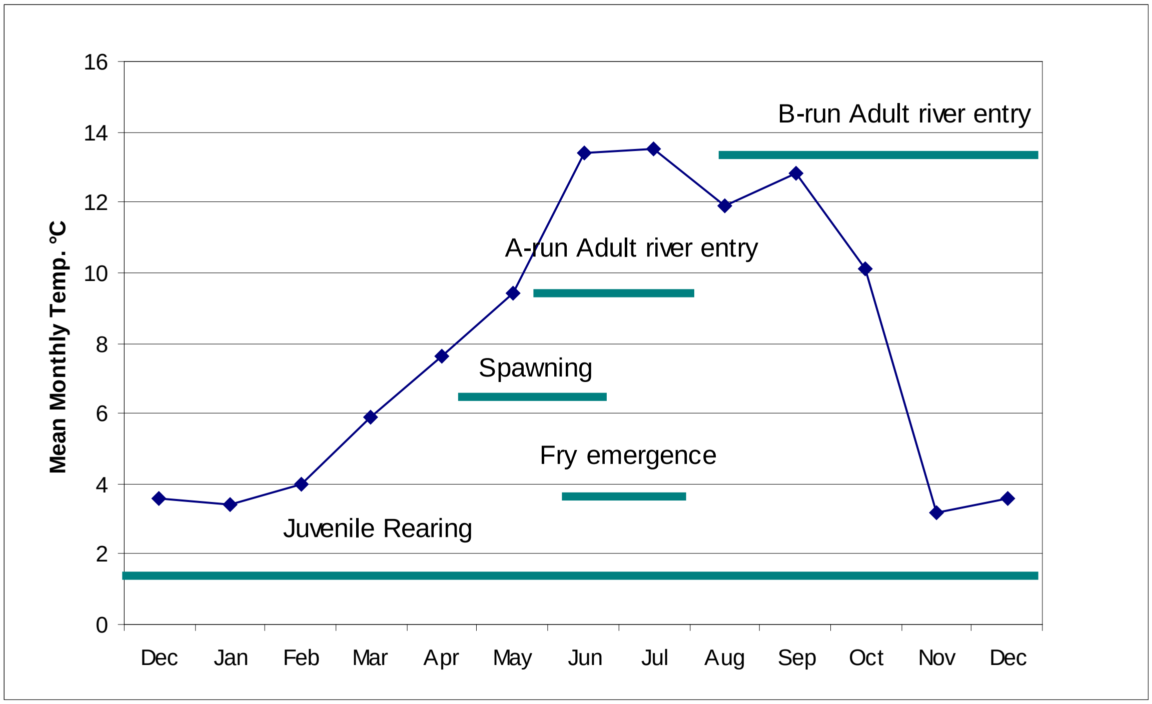

Figure 7. Steelhead life history and mean monthly water temperatures in the Clearwater River, Idaho (Sources: Idaho Department of Fish and Game[20] and USGS Surface-Water Monthly Statistics for the Nation, USGS 13342500 Clearwater River at Spalding ID[21]).

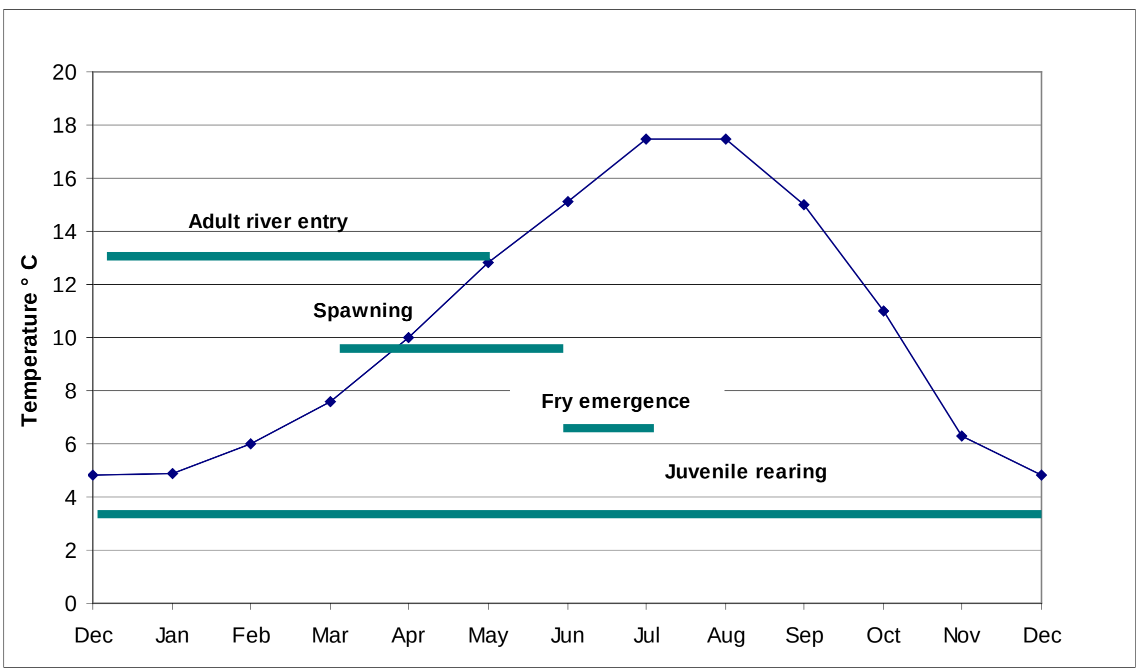

Figure 8. Winter steelhead life-history and mean monthly water temperatures in the Puyallup River Basin, Washington (Ball 2004; and B. Smith, Puyallup Tribe Fisheries, pers. comm., Oct. 14, 2008).

[20] URL:http:fishandgame.adaho.gov/fish/fish_id/steelhead.cfm
[21] URL:http://waterdata.usgs.gov/nwis/monthly?

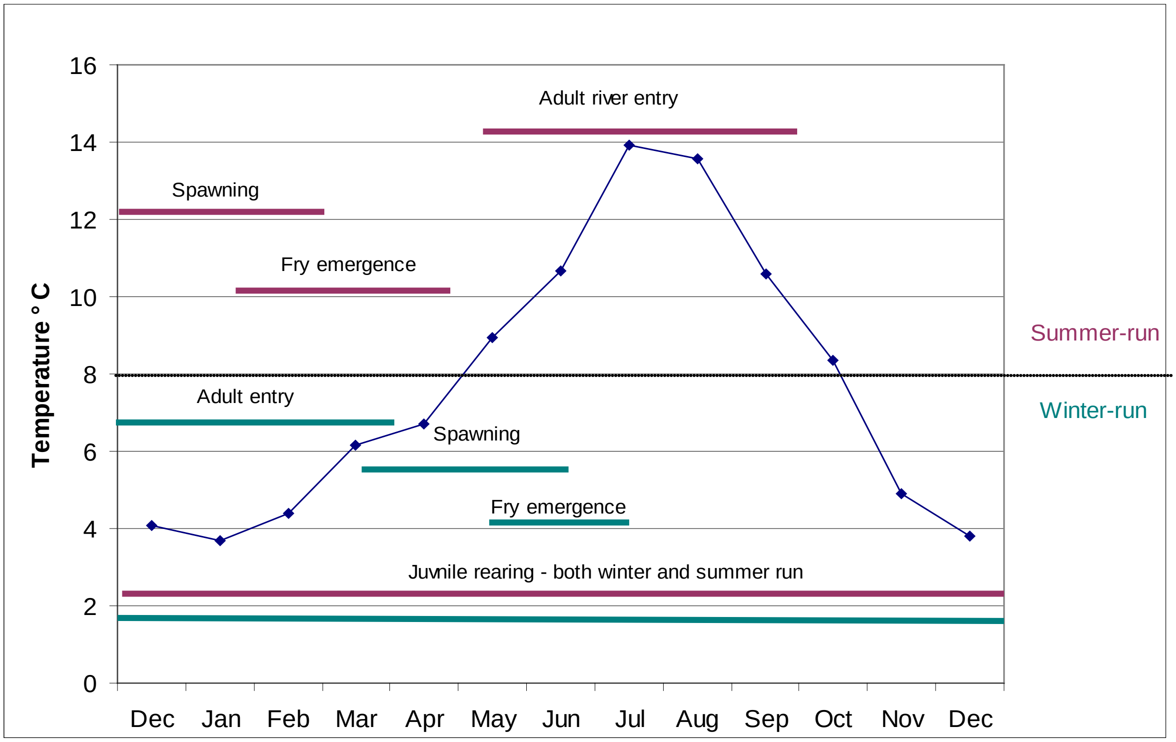

Figure 9. North Umpqua River steelhead life history and average monthly water temperatures (Source: USGS National Water Information System, URL: http://nwis.waterdata.usgs.gov)

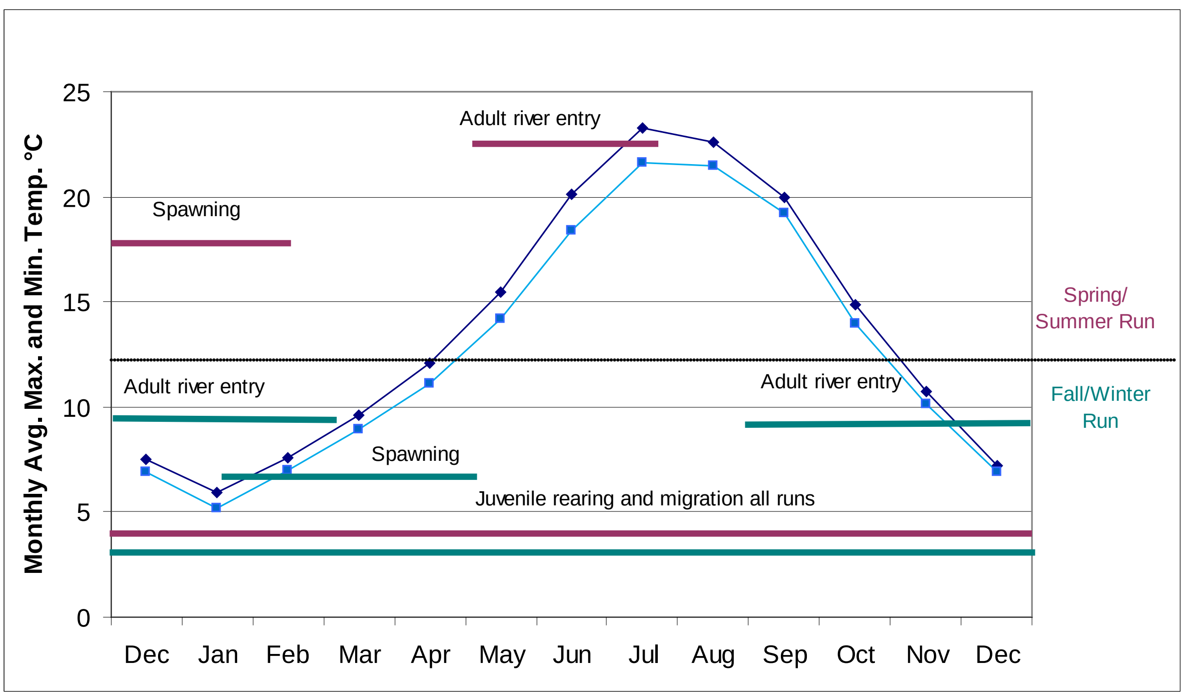

Figure 10. Klamath River steelhead life history and average min. & max. monthly water temperatures (Sources: USFWS 1998 and USGS 2007).

The Influence of Dissolved Oxygen on Tolerance Limits

Generally, in environments where DO is less than optimal fish will compensate for the reduction in DO by increasing gill movement and ventilation volume, in an attempt to maintain adequate oxygen volumes. Cyanide is a powerful asphyxiant, and the addition of cyanide in waters with low DO further stresses fish, reducing the lethal concentration at which survival is typically expected. That cyanide toxicity is influenced by DO is well known (Downing 1954; Smith et al. 1978; Doudoroff 1976; Towill et al. 1978; Alabaster et al. 1983; EPA 1985; Dzombak et al. 2006). Smith et al. (1978) found that a about a 40% reduction in DO levels lead to a 20 to 30% reduction in lethal thresholds for brook trout and rainbow trout. Similarly, Downing (1954) found that rainbow trout survival time increased as DO increased, and the rate of increase did not fall off as DO approach saturation. Alabaster et al. (1983) also demonstrated that the 24-hr LC_{50} value varies with DO concentrations, but not with salinity, and when DO was as low as 3.5 mg/L the LC_{50} value for Atlantic salmon was 24 µg CN/L HCN (well below the acute EC_A reported in Table 4 of the final cyanide BE of 39.65 µg CN/L).

Since the conditions under which this study was conducted are very important to a comprehensive effects analysis, we asked that EPA consider it further in their BE. Unfortunately, EPA responded that (a) it was not considered in the criteria derivation (or was relegated to "other data"), (b) that considerations of toxicity values obtained under a combination of low DO and chemical toxicity is generally not included in their BE because such exposure conditions are not wide-spread across the nation, and (c) confound the toxicity of cyanide alone to which the criteria apply. The first part of the statement, that the data in Alabaster et al. (1983) was not considered in the derivation of the cyanide criteria is interesting, but not necessarily pertinent for the purposes of a Section 7 consultation. First, the criteria were derived without consideration of listed species, but more importantly the question of risk depends upon the environmental decision-making context. That is, Section 7 is first concerned with the risk an action poses individuals of a listed species –this is the level at which a federal agency makes their effect determination. Not until individual effects can be dismissed as insignificant or discountable, would we conclude that an action is "not likely to adversely effect" listed species. The CWA decision-making process begins by focusing not on the individual, but whether community level effects are likely. The effect threshold is considerably different. By the time community level effects are measurable (and noticed) the hazard's risk may pose substantial impacts to small populations. If EPA meant to imply that because a study was not used to derive a particular criterion that it did not warrant consideration in their Section 7 effects analysis, then EPA is missing the point of Section 7 under the ESA. The relevant inquiry is not whether such a study was used by EPA in their 1985 criteria decision, but whether there is information to suggest that environmental conditions to which listed species are exposed may influence the toxicity of the chemical under review—in this case cyanide. Cyanide can be more toxic to freshwater fish at low dissolved oxygen concentrations.

According to our assessment of water quality conditions across the nation, low DO conditions are a problem in many basins at various times of the year (see the Environmental Baseline section of this opinion, also see EPA 2006). The susceptibility of fish to cyanide at low DO may be correlated with the rate of breathing. That is, as a general matter the rate of gill movement increases with decreasing DO, causing the fish to pump additional water through the gills to obtain more oxygen. When cyanide is present in the water column, this may increase the rate at which tissue that is more susceptible to absorption is exposed to cyanide. Although EPA did not consider the relationship between DO and cyanide to be one that should drive the nationally recommended criteria, there is sufficient information indicating the toxicity threshold for salmonids is reduced in low DO conditions such that additional studies are warranted to make definitive conclusions regarding the effect on fish and whether the criteria would fully protect salmonids at the local or site-specific scale.

Effects of Mixtures

Relatively few studies have been performed to measure the effects of free cyanide in combination with other contaminants. Concurrent exposure to cyanide and ammonia produced greater than additive effects to acute lethality in rainbow trout, salmon, and chub (Smith et al, 1979; Alabaster et al., 1983; and Douderoff 1976), and to chronic sublethal effects to growth in rainbow trout (Smith et al 1979). In rainbow trout and salmon, effects to acute lethality were 1.2 and 1.63 times greater than would be expected by additivity. Concurrent exposure to cyanide and zinc also resulted in synergistic effects to acute lethality in fathead minnows, where toxicity was 1.4 times that predicted by additivity (Smith et al 1979). Though we are unable to quantify the effect of these synergistic mechanisms for this analysis, they should be considered when assessing effects of cyanide to aquatic organisms in waterways with elevated concentrations of ammonia and zinc.

Chronic Toxicity

Chronic cyanide toxicity tests have been conducted with relatively few fish species, however, the available data indicate that cyanide not only reduces survival but also affects reproduction, weight gain, growth and development, swimming performance, condition, and development. Few studies have examined the sublethal responses at cyanide concentrations below the freshwater CCC (i.e., <5 µg CN/L) and many have evaluated the effect of concentrations double that of the CCC, making it difficult to evaluate the effect of exposing individuals at the CCC.

Dixon and Leduc (1981) also found evidence of liver necrosis in rainbow trout from low-level exposures of cyanide; however the lowest concentration that they examined was 10 µg HCN/L (~9.8 µg CN/L). In calculating the chronic EC_A value for rainbow trout, it appears that EPA used the reported NOEC from only one study, Dixon and Leduc (1981). In Table 1 of the final cyanide BE, EPA reports a chronic EC_A value for rainbow trout of 9.8µg/L. However, since Dixon and Leduc (1981) did not evaluate rainbow trout response to cyanide concentrations below 9.8 µg/L, it is equivocal to equate this value to a NOEC for the species since adverse effects could not be distinguished at concentrations below this value.

Given the available data reproduction appears to be one of the most sensitive (and most studied) endpoints. Full and partial life cycle tests with fathead minnow and brook trout have shown that fish exposed to sublethal concentrations of cyanide spawned fewer eggs than non-exposed fish (Koenst et al. 1977; Lind et al. 1977). Fecundity was reduced by 57.8% and 46.9% (compared to controls) in female fathead minnows exposed to cyanide at 19.6 µg HCN/L (the LOEC) and 12.9 µg HCN/L (the NOEC), respectively. Similarly, the mean number of eggs spawned by brook trout was reduced by 53.3% at 11.2 µg HCN/L and by 17.7% at 5.7 µg HCN/L. Koenst et al.

(1977) exposed brook trout to nominal HCN concentrations between 5.7 and 77 µg HCN/L, and found that at the mean number of eggs spawned per female decreased with increasing HCN concentrations above 5.7 µg HCN/L. Using a mean temperature of 13.5 ˚C, to convert to CN^- results in a NOEC value is 5.6 µg CN/L, just above the CCC. In the same study, Koenst et al. (1977) found that exposure to 5.5 HCN/L (5.4 µg CN/L) reduced the length of brook trout at hatching and the percentage of eggs that hatched.

Kimball et al. (1978) studied the chronic toxicity of HCN to bluegill and found that bluegill ceased spawning at 5 µg HCN/L (~4.8 µg CN/L). Of eight tests with different concentrations, ranging from 5 to 80.0 µg HCN/L, no spawning was recorded in seven of the tests. Interestingly, at the highest concentration 80.0 µg HCN/L, one female survived and managed to spawn, although her egg production was markedly reduced in comparison to controls. Although the single spawning is difficult to explain, the fact that spawning was completely inhibited in 42 of 43 cyanide-exposed females suggests that bluegill may be particularly sensitive to cyanide at low levels. Results of the tests conducted by Kimball et al. (1978) suggest there was a 3% probability that a female would spawn at $\geq$4.8 µg CN/L, and since levels less than 5.2 µg HCN/L were not tested the only we can only safely conclude that this is a LOEC and that the NOEC lies at a threshold concentration below 5.2 µg HCN/L. Considering the overwhelming evidence of an adverse effect, it is surprising that additional studies on the effects of cyanide on bluegill reproduction have not been conducted over the past 30 years. Cheng and Ruby (1981) studied the effects of pulsed exposures of cyanide on flagfish reproduction. Unlike the studies describe above, where fish were exposed over an extended period of time to a constant concentration, flagfish were exposed to sublethal concentrations of cyanide for 5 day pulses. Flagfish exposed to cyanide (65 µg/L) for 5 days following fertilization, i.e. as eggs, and then reared to maturity in clean water, spawned 25.6% fewer eggs than flagfish that had not been exposed. In another experiment by the same authors, flagfish that received a second 5-day pulse of cyanide as juveniles had an even greater reduction (39.3%) in number of eggs spawned. These studies demonstrate that cyanide can affect an apical reproductive endpoint in fish.

The mechanism by which cyanide induces these reproductive effects is not fully understood, however, key physiological, biochemical, histological (morphological), and endocrine functions known to be involved in sexual maturation are affected by cyanide. For instance, Lesniak and Ruby (1982) reported abnormal oocyte development in sexually maturing female rainbow trout exposed to cyanide (10 and 20 µg HCN/L) for 20 days. Ovaries from cyanide-exposed fish contained fewer mature oocytes, exhibited altered patterns of secondary yolk deposition (in developing oocytes), had nearly twice the frequency of atresia (oocyte resorption), and had an overall reduction in the number of viable eggs. Ruby et al. (1986) reported that vitellogenic female rainbow trout exposed for 12 days to 10 µg HCN/L had lower levels of plasma vitellogenin and a lower gonadosomatic index (GSI) compared to controls. In two similar studies, oocyte diameter (an indicator of gonadal growth and development) was reduced in sexually maturing female rainbow trout exposed for 12 days to 10 µg HCN/L (Ruby et al. 1993a, Szabo et al. 1991). Reduced oocyte diameter was accompanied by reductions in plasma vitellogenin, 17β-estradiol (E2), and GSI (Ruby et al. 1993a), as well as increased whole brain dopamine levels (Szabo et al. 1991).

Dopamine has an inhibitory effect on gonadotropin-releasing hormone (GnRH) neurons in some fish species and it is GnRH which stimulates the release of gonadotropins (GtH I and GtH II) from the pituitary (Patino 1997; Saligaut et al. 1999). GtH I and GtH II are believed to function similar to follicle-stimulating hormone and luteinizing hormone, respectively, in tetrapods (Patino 1997). In female fish, GtH I acts on target cells in the gonad, stimulating E2 synthesis. E2 induces vitellogenin synthesis in the liver. Vitellogenin is the egg yolk precursor in fish which is produced by the liver, transported via blood, taken up by ovaries, and incorporated into developing oocytes. GtH II also acts on the gonad by inducing the synthesis of maturation-inducing steroid (MIS). MIS induces oocyte maturational competence and ovulation (Park et al. 2007; Patino 1997). The control exerted by dopamine over gonadal maturation has been recognized by fish culturists, who have been successful in treating captive-reared fish with anti-dopaminergic drugs (which block dopamine receptors), such as pimozide and domperidone, to induce ovulation (Jensen 1993; Park et al. 2007; Patino 1997; Szabo et al. 2002). Thus, oocyte development, maturation and ovulation are under the control of gonadotropins and E2 which in turn, are modulated in part by GnRH and dopamine. This interaction between the neuroendocrine system and reproductive organs is referred to as the hypothalamus-pituitary-gonadal (HPG) axis (IPCS 2002).

Cyanide has also been shown to affect male reproductive processes. Exposure of male rainbow trout to cyanide (10 and 30 µg HCN/L) for 18 days disrupted spermatogenesis as evidenced by a reduction in the number of dividing spermatogonia and a blockage of mitotic progress (Ruby et al. 1979). Exposure of rainbow trout for 12 days to 10 µg HCN/L resulted in higher numbers of spermatogonial cysts in testes of male trout as well as higher levels of whole brain dopamine (Szabo et al. 1991). Similar results were reported by Ruby et al. (1993) where the number of spermatocytes decreased and the number of spermatocyte precursors (spermatogonial cysts) increased in two-year-old sexually maturing rainbow trout after 12 day exposure to 10 µg HCN/L. There are indications that the transformation of spermatogonial cysts to spermatocytes is hormonally regulated through GtH along the HPG axis and that, within the pituitary, GtH is released from type I granular basophils (Ruby et al. 1993). Histological examination of pituitary glands from cyanide-exposed fish showed a reduction in the number of type I granular basophils. The authors suggested that elevated levels of brain dopamine may be responsible for the selective loss of type I granular basophils and subsequent alteration of spermatocyte formation.

Ruby et al. (1979, 1993) and Szabo et al. (1991) hypothesized that cyanide acts through the HPG axis to affect reproduction in fish. Their studies (described above) demonstrated (1) that cyanide caused an increase in brain dopamine levels, consistent with neuronal effects observed on mammals, (2) that levels of reproductive hormones (E2) and egg-yolk precursors (vitellogenin) were altered following exposure to cyanide, (3) the selective loss of putative GtH releasing pituitary cells (type I granular basophils) and (4) retarded gonad development in cyanide-exposed male and female rainbow trout. Taken together, these results appear to be consistent with HPG axis involvement. In addition, the authors found that these effects occurred following relatively short, sublethal exposures to cyanide (12 – 18 days). Whether these effects would result in the same type of reduced fecundity and spawning, as was observed in cyanide-exposed female fathead minnow (Lind et al. 1977), bluegill (Kimball et al. 1978), and brook trout (Koenst et al. 1977), was not addressed in the rainbow trout studies because they were terminated before the fish reached full sexual maturity, however, it does seem likely. Results from Cheng and Ruby (1981) indicate that continuous exposure to cyanide through the spawning period may not be necessary to affect fecundity. Short-term, pulsed exposures of cyanide to flagfish were sufficient to induce later effects on the number of eggs spawned, and exposed fish did not appear to recover once the exposure had ceased. Even exposure of eggs, one of the most tolerant life stages in terms of acute toxicity (Smith et al. 1979), resulted in latent effects on fecundity once embryos hatched and survived to maturity. Interestingly, it is during early developmental stages that the HPG endocrine axis is set up and feedback sensitivity of the hypothalamus and pituitary gonadotropes to gonadal steroids is established (IPCS 2002). Although Cheng and Ruby (1981) did not measure specific indicators of endocrine axis function, they did find that the pituitary gland of cyanide-exposed flagfish embryos was significantly smaller than the pituitaries from control fish. It would appear that cyanide, like many EDCs (endocrine disrupting compounds, IPCS 2002), may affect the "set up" of the HPG axis and that these early developmental effects may have long term consequences on reproduction.

Chronic exposure of eggs and larvae to cyanide can result in reduced embryo/larvae survival and altered development. Leduc (1978) exposed newly fertilized Atlantic salmon eggs to cyanide (10 - 100 µg HCN/L) and observed teratogenesis, as well as, delayed hatching and reduced hatching success at higher concentrations. There was a dose-dependent increase in the frequency of abnormal fry, ranging from 5.8% to 18.5%. Abnormalities included malformed and/or absence of eyes, defects in the mouth and vertebral column and yolk-sac dropsy (*Hydrocoele embryonalis*, also known as blue sac disease). Similar eye abnormalities were reported by Cheng and Ruby (1981) in flagfish larvae exposed, as eggs, to cyanide (65, 75, 87, 150 µg HCN/L). Hatching success was also reduced and time to hatch was delayed in all cyanide treatments. In a 28-day embryo/juvenile toxicity test, sheepshead minnow survival was significantly reduced in all treatments >29 µg HCN/L (Schimmel 1981). The author noted there was considerable embryonic mortality and that there was no larval mortality during the last two weeks of exposure, indicating a greater sensitivity during early development. Kimball et al. (1978) exposed bluegill eggs and larvae to cyanide (4.8 – 82.1 µg/HCN/L) and reported that most deaths occurred within the first 30 days after hatching. Survival was reduced in all cyanide treatments and the effects were statistically significant at cyanide concentrations >9.1 µg HCN/L.

As previously mentioned, cyanide effects oxidative metabolism, energy production, and thyroid function; all are important for normal growth and performance. Therefore, it is not surprising that sublethal exposure of fish to cyanide has been shown to impact growth, condition and swimming performance. There is also evidence that the effect of cyanide on these physiological endpoints can be modulated by other factors such as diet/ration and temperature. When cichlids (*Cichlasoma bimaculatum*) were fed unlimited rations and exposed to cyanide for 24 days, those fish exposed to lower concentrations of cyanide (< 0.06 µg HCN/L) were larger than controls, where as, at higher concentrations weight gain was depressed (Leduc 1984). The increased weight gain in the low-dose treatments was attributed to higher food consumption, which was allowed to occur because ration was not restricted. Low-dose stimulation is a common effect across a broad range of chemical and non-chemical stressors (Calabrese 2008). Dixon and Leduc (1981) held juvenile rainbow trout on restricted rations and exposed them to cyanide (10, 20, 30 µg HCN/L) for 18 days and observed significantly reduced weight gain in all treatments compared to controls. The effect was characterized by an initial decrease in specific growth during the first 9 days followed by a significant increase from day 9 through 18. The growth surge during the latter half of the exposure period was not sufficient to offset early reductions.

Cyanide-affected fish were in poorer condition, as indicated by lower fat content, and had higher respiration rates for several days post exposure. In addition, fish in all cyanide treatments exhibited degenerative necrosis of hepatocytes, i.e. liver tissue damage, which increased in severity with the level cyanide exposure. Kovacs (1979) held juvenile rainbow trout on restricted rations and exposed them to cyanide for 20 days. The results were similar to those of Dixon and Leduc (1981). Cyanide reduced the mean specific growth rate and affected-fish gained less fat during the exposure period. Kovacs (1979) also examined the effects of temperature on rainbow trout growth and sensitivity to cyanide, and found that the growth rate of rainbow trout was inversely related to holding temperature (6, 12 and 18˚C), as would be expected, and that trout held at colder temperatures were more sensitive to cyanide. The NOECs for mean specific growth rate were 5, 20, and 30 µg HCN/L for trout held at 6, 12, and 18˚C, respectively. Based on the exposure response curves the author estimated thresholds for effects on growth to be <5 µg HCN/L at 6˚C (<4.9 µg CN/L, just below the freshwater CCC), 10 µg HCN/L at 12˚C, and 30 µg HCN/L at 30˚C. In the same study, swimming performance was found to be affected by cyanide and the effect was also temperature-sensitive. Fish from the growth study were placed in swimming chambers and tested for swimming stamina. Among non-exposed trout, swimming stamina, measured as distance travelled (meters), decreased with decreasing temperature, i.e. fish held a 6˚C travelled a shorter distance than fish held at 18˚C. Cyanide-exposed fish had reduced swimming stamina compared to non-exposed fish and the effect was more severe at colder temperatures. Based on the exposure-response regression equations reported by Kovacs (1979) the predicted reduction in swimming stamina (compared to controls) for fish exposed to cyanide at the chronic water quality criterion (5.2 µg CN/L) would be 52% at 6˚C, 20% at 12˚C, and 3% at 18˚C. Several other authors have studied swimming performance as well. Leduc (1966) studied the effect of sublethal concentrations of cyanide on cichlids and coho salmon; the lowest concentration examined was 7 µg CN/L. At 7 and 8 µg CN/L cichlids exhibited reduced swimming speeds, similar to fish exposed to higher concentrations (30 µg CN/L; Leduc 1966). Neil (1957 in Kovacs 1979, Koenst et al. 1977) also showed that cyanide concentrations as low as 10 µg CN/L reduced the swimming stamina of brook trout by 98%. Similarly Broderius (1970) and Speyer (1975) observed reduced the swimming ability of coho salmon and rainbow trout at concentrations of 10 and 20 µg HCN/L. Thus, chronic exposure of fish to cyanide at sublethal concentrations, can affect growth, condition and swimming performance, and factors such as temperature and diet/ration can modulate cyanide toxicity. Neither fat synthesis nor swimming performance, however, are endpoints that EPA would typically use to establish water quality criteria, yet the two endpoints can significantly influence an individual's fitness. Fat is an indicator of growth, and is important during migration and reproduction as an energy reserve. Poor swimming performance can reduce ability to escape predators, maintain stream position, migratory performance. That adverse effects occur below the CCC appears unequivocal; a question that merits further investigation is just how far below the CCC is the threshold response for most species?

Chronic Effects Estimation

Ideally, we would use concentration (dose)-response data to build predictive models of the potential sublethal effects of cyanide. Unfortunately, such data do not exist for cyanide or listed species. As recently reviewed by Gensemer et al. (2007), the current inventory of concentration-response data from chronic toxicity testing with cyanide consists of four datasets; one each for reproductive endpoints among fathead minnow (*Pimephales promelas*; Lind et al. 1977) and brook trout (*Salvelinus fontinalis*; Koenst et al. 1977); and for juvenile survivorship among bluegill (*Lepomis macrochirus*; Kimball et al. 1978) and sheepshead minnow (*Cyprinodon variegates*; Schimmel et al. 1981). Upon closer inspection, Gensemer et al. (2007) found the dataset for sheepshead minnow to be insufficient for meaningful predictive modeling and we agree with that conclusion. Thus, we are left with three datasets as the best available scientific basis for estimating toxic effects at the chronic criterion value of 5.2 µg CN/L. In addition to our three useable concentration-response datasets we also possess estimates of LC_{50} values for our listed species as per the procedures established in the *Methods Manual*.

To estimate the chronic effects of the proposed action on listed species, we transformed our three concentration-response data sets into the most precise predictive concentration-response models that the data can support and then used these models to predict the response of chronic toxicity test species to the CCC for cyanide. We assume that the predicted response of a listed fish species to the CCC is the same as the response observed for a chronic toxicity test species at an adjusted chronic CN exposure level based on the ratio of their respective LC_{50} values (example below). This approach is based on two simplifying assumptions:

1. That relative differences in sensitivity to chronic CN exposures between our listed evaluation species and our chronic toxicity test species (i.e., fathead minnow, brook trout, and bluegill) are approximated by the ratio of their respective LC_{50} values, and

2. The slopes of the concentration-response curves are also approximately comparable between our listed evaluation species and our chronic toxicity test species.

These assumptions create a clearly defined basis for a default hypothesis that allows us to proceed within the constraints of minimal data until such time as more data become available. As more data become available appropriate modification (or validation) of our default approach is necessary.

To provide an illustrative example of the outcome from our simplifying assumptions, suppose that one chronic toxicity test species is predicted to exhibit a 20% adverse effect from 5.2 µg CN/L. If a listed species happens to have an estimated LC_{50} value equal to that of the chronic toxicity test species, then a 20% adverse effect would also be predicted for the listed species. If the ratio of LC_{50} values was 1.5 (rather than 1.0) in the direction of greater sensitivity for the listed species than the chronic toxicity test species, then the predicted response at the concentration of interest of 5.2 µg/L for the listed species would be the same as the response observed for the chronic toxicity test species at a CN concentration 1.5 times 5.2 µg/L, i.e., at 7.8 µg/L. We refer to such surrogate currency equivalents for our listed species as $SSEC_x$ values (or sometimes shortened to SS_x). In this example, the predicted adverse effect for chronic toxicity test species at the $SSEC_x$ of 7.8 µg/L would be our surrogate currency predicted effect for the listed species at 5.2 µg CN/L (from that one of three prediction models) for the purposes of this Opinion. A more detailed derivation and explanation of the $SSEC_x$ concept is provided in Appendix D.

Because groups of taxonomically related listed species were assigned identical LC_{50} values from the same ICE model, there are only 17 $SSEC_x$ values that need to be evaluated for any given (chronic toxicity test species) prediction model, but they are different for each prediction model (3 x 17 = 51 total $SSEC_x$ values of interest). For the prediction model based on fathead minnow chronic toxicity data the $SSEC_x$ values range from 6.7 to 45.8 µg CN/L (Table 40). As indicated by the entire range of $SSEC_x$ values being greater than 5.2 µg CN/L, all listed evaluation species have LC_{50} values that are more sensitive to cyanide than the fathead minnow LC_{50} value. For the prediction model based on brook trout chronic toxicity data the $SSEC_x$ values range from 4.2 to 28.4 µg CN/L (Table 40). For the prediction model based on bluegill chronic toxicity data the $SSEC_x$ values range from 6.1 to 41.7 µg CN/L (Table 40). Those $SSEC_x$ ranges define for each prediction model the range of cyanide concentrations over which model fit will be of most relevance to this Opinion. Detailed $SSEC_x$ results and the origins of the LC_{50} values used to calculate the $SSEC_x$ values are presented in Table 40 and Appendix D.

Table 43. Surrogate currency equivalents ($SSEC_x$[1]) for each LC_{50} surrogate taxon/chronic toxicity test species combination

Surrogate taxa used to estimate listed species (LS) LC_{50}	**$LSEC_x$ (µg CN/L)**	**LS LC_{50} (µg CN/L)**	**Effects on Fecundity**		**Effects on Early Life Stage Survival**
			Fathead Minnow SS LC_{50}=138.4 (µg CN/L)	**Brook Trout SS LC_{50}=85.7 (µg CN/L)**	**Bluegill SS LC_{50}=126.1 (µg CN/L)**
			$SSEC_X$ (µg CN/L)	**$SSEC_X$ (µg CN/L)**	**$SSEC_X$ (µg CN/L)**
Actinopterygii (class**)**	5.2	66.5[2]	10.8	6.7	9.9
Cypriniformes (order)	5.2	84.55[2]	8.5	5.3	7.8
Family Catostomidae					
Xyrauchen texanus (species)	5.2	83.8[3]	8.6	5.3	7.8
Family Cyprinidae					
Cyprinella monacha (species)	5.2	36.4[3]	19.8	12.2	18.0
Gila elegans (species)	5.2	50.9[3]	14.1	8.8	12.9
Notropis mekistocholas (species)	5.2	48.5[3]	14.8	9.2	13.5
Ptychocheilus lucius (species)	5.2	43.5[3]	16.6	10.3	15.1
Perciformes (order)	5.2	90.8[2]	7.9	4.9	7.2
Percidae (family)	5.2	42.3[3]	17.0	10.5	15.5
Etheostoma (genus)	5.2	40.0[3]	18.0	11.1	16.4
Etheostoma fonticola (species)	5.2	21.5[3]	33.4	20.7	30.5
Order Salmoniformes, Family Salmonidae					
Oncorhynchus (genus)	5.2	47.0[3]	15.3	9.5	13.9

Oncorhynchus apache (species)	5.2	16.5[3]	43.6	27.0	39.7
Oncorhynchus tschawytscha (species)	5.2	32.0[3]	22.5	13.9	20.5
Oncorhynchus kisutch (species)	5.2	32.4[3]	22.2	13.8	20.3
Oncorhynchus clarki henshawi (species)	5.2	22.8[3]	31.5	19.5	28.7
Salvelinus (genus)	5.2	15.7[3]	45.8	28.4	41.7
Salmo salar (species)	5.2	90[4]	8	5	7.3

[1] $SSEC_x$ values were calculated using equation 5 in Appendix D. Surrogate taxa were used to estimate LC_{50} values for listed species.
[2] LC_{50} based on 5th percentile estimate from species sensitivity distribution (SSD), Table 2 – Cyanide BE.
[3] LC_{50} estimate based on lower bound of the 95% CI from ICE model
[4] LC_{50} based on measured value from the Cyanide BE

Prediction models. We applied statistical regression techniques to model, or "fit", the relationship between cyanide concentrations and toxic effects based on data for our chronic toxicity test species. For nuances of statistical regression specific to toxicological applications we relied substantively on two recent technical guidance documents: (1) Environment Canada (2005: "Guidance Document on Statistical Methods for Environmental Toxicity Tests"), and (2) OECD (2006: "Current Approaches in the Statistical Analysis of Ecotoxicity Data: A Guidance to Application"). We also reviewed other relevant guidance such as that provided in the documentation for EPA's Toxicity Relationship Analysis Program (TRAP) (EPA 2002) and in discipline-specific statistical textbooks such as Gad and Weil (1988) and Sparks (2000).

As noted by Environment Canada (2005) an important principle of regression techniques is to keep the model simple, if that can reasonably be done. We have further incentive to follow that principle because we have a strong interest in evaluating the uncertainty (confidence) associated with point estimates and therefore an interest in avoiding what Environment Canada (2005) noted as the "...obstacle of calculating confidence intervals around nonlinear regression estimates..." Throughout this exercise we have been mindful that because our models are not based on biological or chemical mechanisms of action, but are purely statistical constructs, they have no biological meaning. A statistical concentration-response model only serves to smooth the observed concentration-response, to estimate effect concentrations by interpolating between treatment concentrations, and to provide a tool for assessing confidence intervals. Therefore the choice of model is to some extent arbitrary (OECD 2006). That being noted, we constructed models that conformed to the data we are working with and with statistical standard practices (such as data transformations). The degree of model fit achieved is an artifact of those specific decisions not the result of *post hoc* "model shopping" (EPA 2002).

Generic concentration-response relationship. Figure 11 illustrates a generic concentration-response relationship which typically takes on a sigmoidal form due to threshold effects on the low concentration end of the x-axis and to asymptotic effects at the high concentration end of the x-axis.

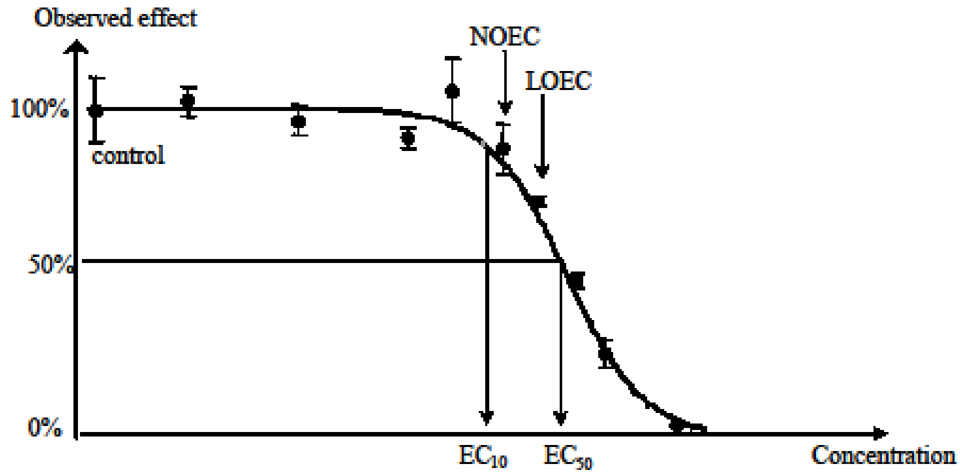

Figure 11. Generalized concentration-response relationship adapted from OECD (2006:Figure 3.2). Note that the illustrated curve is a plot fitted to a real dataset, thus the identification of NOEC and LOEC concentrations. For the purposes of this figure think of the y-axis as a positive attribute that becomes diminished by toxicity, such as percent survivorship.

Note that the superimposed straight line in Figure 11 represents the region of concentrations that induce an intermediate toxic response that are well approximated by a linear fit. This "linear region" is strongest within one probit (also known as "normal equivalent deviate") either side of the median response concentration (EC_{50}), or roughly for concentrations that induce 16 to 84% response (Environment Canada 2005). The narrow ranges of $SSEC_x$ values that we need to evaluate can be expected to overwhelmingly fall within those boundaries as a result of the methods EPA used to set the chronic criterion at 5.2 µg CN/L (see the next section titled, *Derivation of the Criterion Continuous Concentration*). Our approach is conceptually similar to the TRAP program's Piecewise Linear regression option (EPA 2002). Even with regard to the nonlinear regression options in TRAP, EPA (2002) provides a recommendation for segmented analysis when there is a focal region (or subset) of test concentrations of particular concern:

> *Within the limitations of this program, one useful approach can be to exclude (censor) high effects data from the analysis if (a) only low levels of effect are of interest and (b) there are sufficient low-to-moderate effects data to support a good analysis.*

Prediction model based on fathead minnow dataset. Lind et al. (1977) examined fathead minnow fecundity (number of eggs per spawn) and egg hatchability in relation to a series of cyanide treatments (concentrations). The experimental structure and fecundity results are summarized in Table 41. There were five control replicates, and two replicates each for ten exposure concentrations. The response data are reasonably monotonic, especially within the intermediate response range covered by the lowest six treatments. Those treatments range (on a free cyanide basis) from 6 to 45.6 ug/L CN; a span that closely corresponds to the $SSEC_x$ range we want to evaluate (Table 4).

Table 44. Egg production of adult fathead minnows exposed for 256 days (from larvae through adult) to various concentrations of cyanide (from Lind et al. 1977; Table II).

Treatment HCN (µg/L)	**Mean HCN (µg/L)**	**Free cyanide as CN (µg/L)**	**Mean eggs per female**	**Mean eggs per female per treatment**	**Reduction in the number of eggs per female - percent of control**
Control			2530	3476	
Control			4483		
Control			3990		
Control			2718		
Control			3660		
5.7	5.8	6.0	1886	2512	27.7
5.9			3138		
13.0	12.9	13.3[N]	1701	1845	46.9
12.7			1989		
19.6	19.6	20.2[L]	1694	1468	57.8
19.6			1241		
27.1	27.3	28.2	1093	1367	60.7
27.5			1640		
36.0	35.8	36.9	678	1010	71.0
35.6			1341		
43.7	44.2	45.6	2054	1124	67.7
44.7			194		
62.5	63.5	65.6	74	72	97.9
64.5			70		
73.1	72.8	75.1	573	319	90.8
72.4			64		
81.5	80.7	83.3	266	243	93.0
79.8			219		
96.1	100.8	103.9	0	0	100.0
105.4			0		

[N]NOEC
[L]LOEC

To “build” our prediction model we transformed both the concentration data and the fecundity data for *a priori* reasons. We log-transform the concentration data for two reasons: (1) statistically, toxicological tolerance distributions have long been confirmed as log-normal (OECD 2006), and (2) biologically, organisms experience toxicants on a log scale. Toxicological custom is to use log base-10 for the log transformations of test concentrations (Environment Canada 2005). Count data, such as “number of eggs per spawn” typically conform to a Poisson distribution rather than a normal distribution. To normalize such data for regression analysis a square-root transformation is recommended (EPA 2002). Thus, we use the square-root transformed response data for statistical analysis and then back-transform for reporting results. This transform does not change the model, but affects what the best parameter estimates and confidence limits are (EPA 2002). Thus, our model of choice is a log-square root linear regression over our focal segment (subset) of test concentrations.

In agreement with Gensemer et al.’s (2007) treatment of the same dataset, we collapse the fecundity and egg hatchability endpoints into a single endpoint, “eggs hatched per spawn” which is the product of (eggs per spawn) x (egg hatchability) at each treatment concentration. We went a step further than Gensemer et al. (2007) and additionally apply a data smoothing procedure to meet the assumption of monotonicity of response inherent in a linear regression. We did that by calculating three-point moving averages for both the fecundity and hatchability endpoints. This is a standard statistical technique for separating the "signal" from the "noise" in epidemiological and earth sciences (e.g., Borradaile 2003; Rothman et al. 2008). Although we didn't use the control data in our focal segment linear regression, we estimate where the smoothed data would cross the y-axis by double-weighting the control value, which then along with its nearest neighboring data point provided the basis of a three-point moving average for the "endpoint" of the concentration series. This double-weighting is justified conceptually because a treatment to the left of the controls on the concentration axis would be expected to respond the same as the controls (Environment Canada 2005). This enables us to avoid comparing point estimates of eggs hatched per spawn from models fitted to smoothed data with "unsmoothed" control reference points. Note that our "smoothed" estimate of a control reference point is obtained using the actual data nearest to the y-axis and is not extrapolated from our estimated regression equation. Also note that we do not control-adjust the results prior to model fitting, a practice that leads to serious upward bias in EC_x point estimates (Environment Canada 2005; OECD 2006). A summary of response data smoothing and transformation is presented in Table 42.

Table 45. Fathead minnow input data for effects modeling

Treatment (free µg CN/L)	Pooled mean eggs/female	Pooled Proportion Hatch[a]	Unsmoothed Pooled mean hatch/female[b]	3-pt moving average of proportion hatch	Smoothed Pooled mean hatch/female[b]	SQRT transform
Control Mean	3476	0.842	2927	0.763[c]	2652	51.5
6.00	2512	0.606	1522	0.754	1894	43.52
13.30	1845	0.813	1500	0.682	1258	35.47
20.20	1468	0.626	919	0.612	898	29.97
28.20	1367	0.396	541	0.527	720	26.83
36.90	1010	0.559	565	0.354	358	18.92
45.60	1124	0.108	121	0.271	305	17.46
65.60	72	0.147	11	0.149	11	3.31
75.10	319	0.192	61	0.181	58	7.62
83.30	243	0.204	50	0.132	32	5.66
103.90	0	0	0	0.068[c]	0	0

[a]Means weighted by replicate sample sizes; excludes hatchability result for Control B as per authors' (Lind et al. 1977:264-265) recommendation
[b]Rounded to the nearest whole number
[c]Based on double-weighted observed value; assuming any doses to the left of 0% response will be constant and any points to the right of 100% response will be constant
[d]Final effects model based upon the shaded subset of data

The resulting log-square root focal segment linear regression model shows a very close fit to the data with an adjusted r-square of 0.964. The regression equation is:

Square-root (hatched eggs per spawn) = -30.19(LOG CN) + 68.36

The regression plot (Figure 12) and summary regression statistics (Table 43) are presented below. The regression was conducted using the multiple linear regression module of the *Statistica* software package (StatSoft 2006). Because we are dealing with small samples, i.e., six points in this case, we report the adjusted r-squared value which adjusts for the limited degrees of freedom in the model (StatSoft 2006).

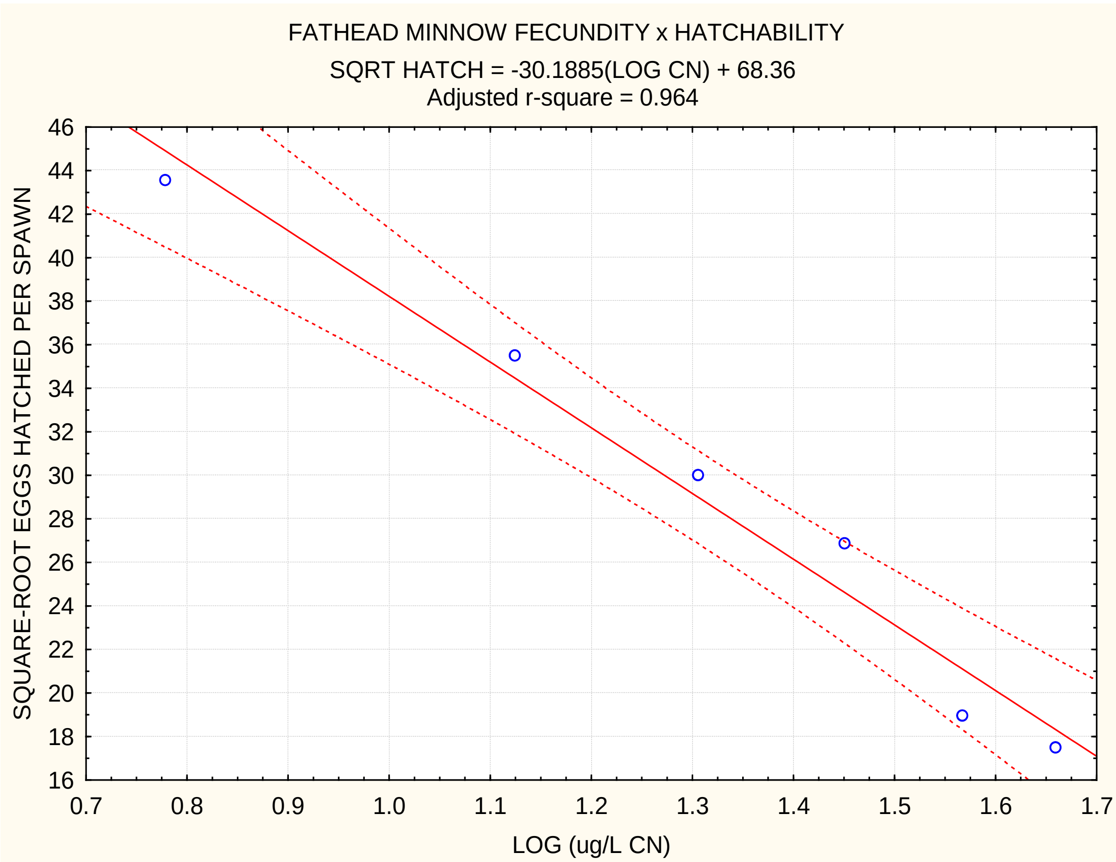

Figure 12. Log- Square Root Focal Segment Regression Plot for Fathead Minnow Fecundity x Hatchability (= Eggs Hatched Per Spawn)

Table 46. Summary regression statistics

Effects Surrogate	N	F value	p-level	Intercept	Std Err	p-level	Slope	Std Err	p-level
Fathead Minnow	6	134.6	<0.00032	68.36	3.505	0.000041	-30.19	2.602	0.00032
Brook Trout	5	12.34	<.039	24.85	2.595	0.0024	-6.594	1.877	0.039
Bluegill	5	11.75	<0.042	0.3514	0.9277	0.73	-2.533	0.7919	0.042

Prediction model based on brook trout dataset. Koenst et al. (1977) examined brook trout fecundity (number of eggs per spawn) and egg viability in relation to a series of cyanide treatments (concentrations). The experimental structure, as well as the fecundity results are summarized below (Table 44). There were two control replicates, and seven cyanide treatments. The lowest five treatments produced intermediate effects responses and covered a range of concentrations from 5.6 to 53.2 µg/L CN; a span that closely corresponds to the $SSEC_x$ range we want to evaluate (Table 40). There was substantive variability in the results for the two control replicates. This lead Koenst et al. (1977) to exclude control replicate B, but noting that additional testing might indicate that the control results should be averaged. As noted in the footnote to Table 44, subsequent studies with brook trout (Holcombe et al. 2000) have confirmed that control replicate B should be averaged with control replicate A and therefore we use the control mean as our reference point for evaluating model predictions.

Table 47. Egg production of adult brook trout exposed to HCN for 144 days prior to the start of spawning (from Koenst et al. 1977)

HCN (µg/L)	Free cyanide as CN (µg/L)	Mean eggs spawned per female	Reduction in the number of eggs per female - percent of control*
Control A		502	
Control B		744	
Control Mean		623	
5.7	5.6	513	17.7
11.2	11.1	291	53.3
32.3	31.9	246	60.5
43.6	43.1	442	29.1
53.9	53.2	262	57.9
64.9	64.1	124	80.1
75.3	74.4	0	100.0

* Reductions in the number of eggs spawned relative to controls were calculated using the Control mean (623 eggs per female). Koenst et al. 1977 performed the same calculation using only Control A (502 eggs per female) and reported that the MATC (Maximum Acceptable Toxicant Concentration) lies between 5.7 and 11.2 µg HCN/L. However, the authors went on to say that "When compared to the mean of the two controls, 5.7 µg/L HCN would appear to show a substantial reduction in eggs spawned per female, but due to the high variability in spawning in the two controls, further study would be required to reach this conclusion." Since that time other studies with brook trout have been conducted (Holcombe et al. 2000). The mean number of eggs spawned per female observed by Koenst et al. 1977 is within the range reported for these other studies, which supports the use of data from both controls in estimating the effect of cyanide on brook trout fecundity.

Again, in agreement with Gensemer et al.'s (2007) treatment of the same dataset, we collapse the fecundity and egg viability endpoints into a single endpoint, "viable eggs per spawn" which is the product of (eggs per spawn) x (egg viability) at each treatment concentration. In the five-point segment of the data that we focus on, there was a substantive deviation from monotonicity at the 43.1 µg/L CN concentration. Therefore, once again we employed data smoothing with a 3-point moving average to restore a monotonic progression of responses. Because the endpoint here is virtually the same as the endpoint for the fathead minnow dataset, other aspects of our treatment of the data for "building" a prediction model are the same as already presented above. A summary of response data smoothing and transformation is presented in Table 45 below.

Table 48. Brook trout input data for effects modeling

Treatment (free CN µg/L)	Mean eggs/female	3-pt moving average of mean eggs/spawn	Proportion Viable	3-pt moving average of proportion viable	Smoothed mean viable/female[a]	SQRT transform
Control Mean	623	586[b]	0.935	0.923[b]	541	23.26
5.60	513	476	0.899	0.872	415	20.37
11.10	291	350	0.781	0.803	281	16.76
31.90	246	326	0.729	0.792	258	16.06
43.10	442	317	0.866	0.745	236	15.36
53.20	262	276	0.641	0.502	139	11.79
64.10	124	129	0	0.214	28	5.29
74.40	0	41[b]	0	0[b]	0	0

[a]Rounded to the nearest whole number
[b]Based on double-weighted observed value; assuming any doses to the left of 0% response will be constant and any points to the right of 100% response will be constant
[c]Final effects model based upon the shaded subset of data

The resulting log-square root focal segment linear regression model does not show as strong a fit to the data as the fathead minnow model does, but still shows a reasonably good fit with an adjusted r-square of 0.739. The regression equation is:

$$\text{Square-root (viable eggs per spawn)} = -6.594(\text{LOG CN}) + 24.85$$

The regression plot is presented in Figure 13 and summary regression statistics are presented in Table 43. The regression was conducted using the multiple linear regression module of the *Statistica* software package (StatSoft 2006). Because we are dealing with small samples, i.e., five points in this case, we report the adjusted r-squared value which adjusts for the limited degrees of freedom in the model (StatSoft 2006).

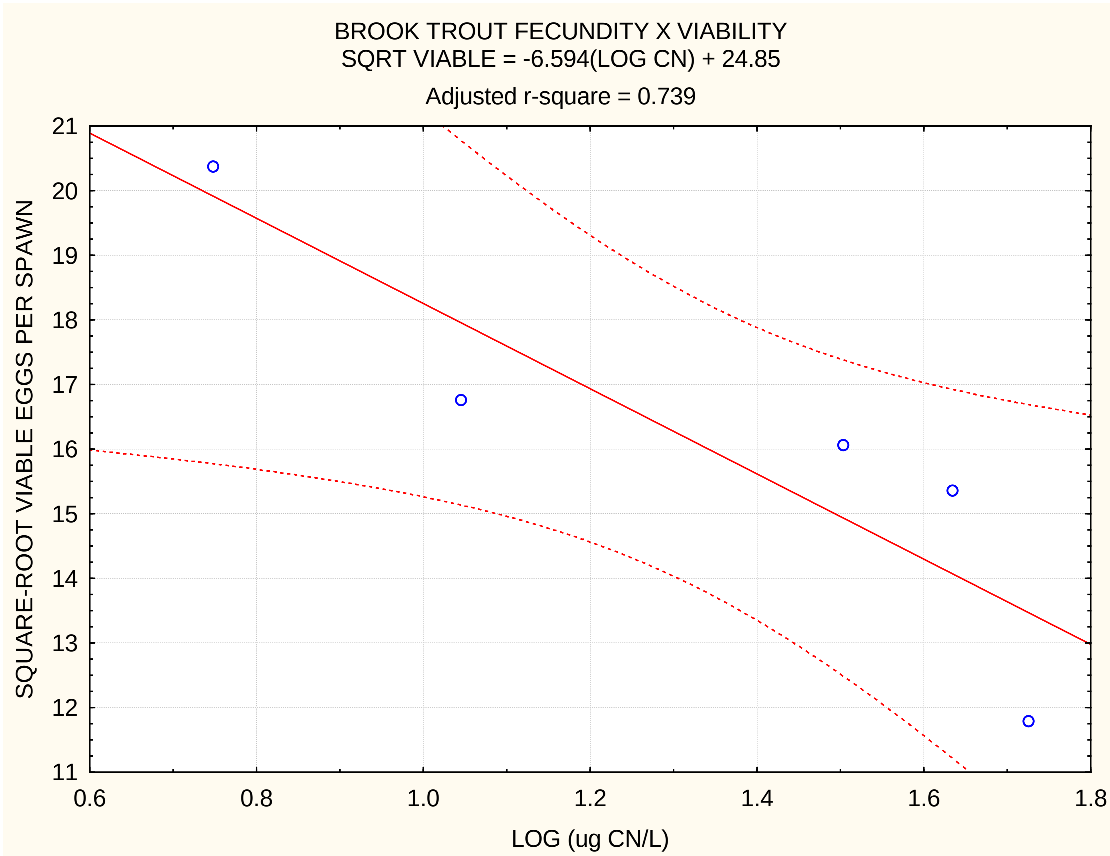

Figure 13. Log-Square Root Focal Segment Regression Plot for Brook Trout Fecundity x Viability(= Viable Eggs per Spawn)

Prediction model based on bluegill dataset. Kimball et al. (1978) examined bluegill juvenile survivorship in relation to a series of cyanide treatments (concentrations). The experimental structure, as well as the survivorship results are summarized in Table 46. There were four control replicates, and two replicates each for eight cyanide treatments. The lowest five treatments produced intermediate effects responses and covered a range of concentrations from 4.9 to 40.6 µg/L CN; a span that closely corresponds to the $SSEC_x$ range we want to evaluate (Table 40).

Table 49. Survival of bluegill from fertilized egg to the 57-day juvenile state in various HCN concentrations (from Kimball et al. 1978)

HCN (µg/L)	**Mean HCN (µg/L)**	**Free cyanide as CN (µg/L)**	**Percent survival**	**Number of surviving juveniles ***	**Mean percent survival**	**Reduction in survival compared to controls**
Control			37.5	75	23.3	
Control			20.0	40		
Control			10.0	20		
Control			25.5	51		
4.8	4.8	4.9	18.5	37	18.5	20.6%
5.2			lost			

8.9	9.1	9.4[N]	25.0	50	16.3	30.0%
9.2			7.5	15		
19.2	19.4	19.9[L]	3.0	6	2.8	88.0%
19.6			2.5	5		
28.5	29.1	29.9	2.5	5	2.5	89.3%
29.7			2.5	5		
38.7	39.5	40.6	3.0	6	3.8	83.7%
40.2			4.5	9		
49.3	49.3	50.7	13.5	27	13.5	42.1%
51.9			lost			
61.8	62.9	64.6	0.0	0	0.0	100.0%
64			0.0	0		
80.4	82.1	84.4	0.0	0	0.0	100.0%
83.8			0.0	0		

*Number of surviving juveniles was calculated by multiplying the reported percent survival times the starting number of fertilized eggs per treatment (200).
[N]NOEC
[L]LOEC

The bluegill dataset differs qualitatively from the fathead minnow and brook trout datasets because the response variable, juvenile survivorship is a quantal (binary) rather than continuous variable. Quantal variables conform to a binomial distribution. Such data are typically analyzed via either probit transformation, as employed by Gensemer et al. (2007), or logit transformation of the proportions of responding and non-responding test subjects. Probits are normal equivalent deviates and logits are logistic equivalent deviates. These two transforms usually yield similar estimates of EC_{50} values, but differ appreciably in their EC estimates in the tails of the distributions.

Environment Canada (2005) recommends logistic methods over probits for "... mathematical simplicity and other good reasons." Logit = ln (p/1-p), where p is the proportion of effected test subjects (e.g., if juvenile survival were 30% for a particular treatment concentration, p would equal 0.3 and the logit transform would equal -0.8473). The logit transform linearizes the sigmoidal logistic response curve (Environment Canada 2005; StatSoft 2006). Furthermore, in fitting the logit model, the control observations can be excluded, as they do not provide any information, unless a background parameter in included (OECD 2006).

Both Environment Canada (2005) and OECD (2006) note that it is common practice to correct the data for background response prior to analysis (for example via Abbott's correction), but that such pre-treatment of the data is unsound statistical practice that can result in substantive overestimation of EC_x values. The bias increases as the control effect being adjusted for increases. We fit a focal segment of the bluegill dataset to a log-logit regression using results that are not control-adjusted prior to analysis. Thus, our prediction model yields unbiased estimates of proportion effect that can be control-adjusted for reporting purposed after the fact. The dataset is reasonably monotonic until the highly anomalous result for the treatment at a concentration of 50.7 µg/L CN. Gensemer et al. (2007) censored that point as an outlier. Because our $SSEC_x$ range extended up to only 41.7 µg/L CN (Table 40) the 50.7 µg/L CN treatment did not fall within our focal segment of concern. The last three treatments in our focal segment produced results of greater than 84% effect which would place them in the nonlinear upper tail of the sigmoidal curve (Figure 11), but unlike a log-square root regression the logit transform will linearize points in the tails relative to intermediate effects points. Thus, for log-logit regression points that fall in tails do not have to be avoided in order to apply linear regression. The minor deviation from monotonicity in the last two points of our focal segment did not warrant data smoothing. A summary of the logit transformed response data is presented in Table 47.

Table 50. Bluegill input data for effects modeling

Treatment (free CN µg/L)	Mean surviving juveniles	Proportion Survival	Logit Proportion Survival
Control Mean	46.5	0.2325	-1.1942
4.9	37	0.1850	-1.4828
9.4	32.5	0.1630	-1.6361
19.9	5.5	0.0280	-3.5472
29.9	5	0.0250	-3.6636
40.6	7.5	0.0380	-3.2314
50.7	27	0.1350	-1.8575
64.6	0	0.0000	
84.4	0	0.0000	

[a]Final effects model based upon the shaded subset of data

The resulting log-logit focal segment linear regression model does not show as strong a fit to the data as the fathead minnow model does, but with an adjusted r-square of 0.729 shows a reasonably good fit comparable to that achieved for the brook trout dataset. The regression equation is:

Logit (proportion juvenile survival) = -2.533 (LOG CN) + 0.3514

The regression plot is presented in Figure 14 and summary regression statistics are presented in Table 43. The regression was conducted using the multiple linear regression module of the *Statistica* software package (StatSoft 2006). Because we are dealing with small samples, i.e., five points in this case, we report the adjusted r-squared value which adjusts for the limited degrees of freedom in the model (StatSoft 2006).

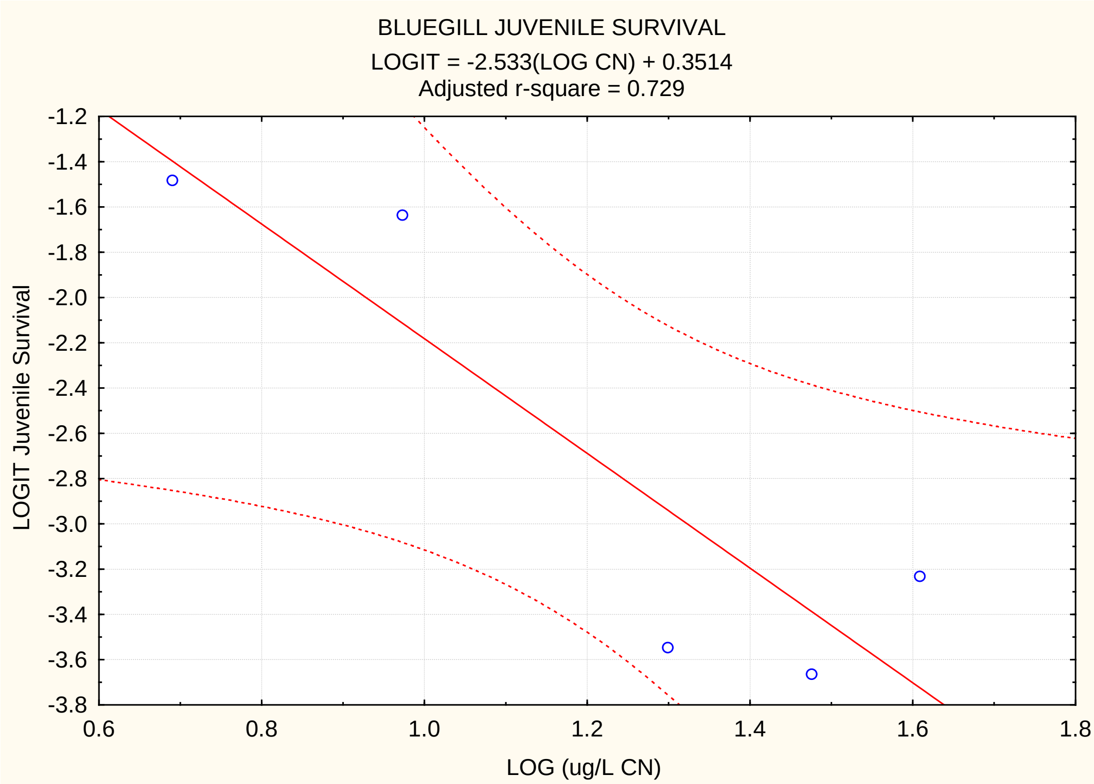

Figure 14. Log-logit focal segment regression plot for bluegill juvenile survival

Prediction Results

Effects predictions are generated by substituting LOG ($SSEC_x$) for LOG (CN) into the prediction regression equations. This was accomplished via the "predict dependent variable" algorithm in the multiple linear regression module of Statistica (StatSoft 2006). That algorithm also uses the estimated standard error of the regression coefficient to generate 95% confidence limits for the predicted point estimates (maximum likelihood estimates). For the fathead minnow and brook trout prediction regressions, the prediction and confidence limit output are in the form of square-roots of numbers of eggs. To convert those predictions to a percent effect, the predicted results were first squared and then scaled for percent change compared to the applicable smoothed control value according to the formula:

$$\% \text{ Effect} = [1- (\text{predicted egg count} / \text{smoothed control value})] \times 100$$

Any predicted egg counts exceeding the smoothed control value were automatically converted to 0% effect. For the bluegill prediction regression, the prediction and confidence limit output are in the form of logit transforms for proportions of juvenile survivorship. The logit transforms are back-transformed to proportions by the formula:

$$\text{Proportion survival} = e^{(\text{logit})} / 1 + e^{(\text{logit})}$$

The predicted survival proportions are scaled for percent change compared to the reported control value according to the formula:

% Effect = [1- (predicted proportion survival / mean control proportion survival)] x 100

Again, any predicted survivorship exceeding the observed mean control survivorship results in a percent effect prediction that is automatically converted to 0% effect. The raw input and output data for effects predictions are presented in Appendix E.

A summary of predicted effects and their estimated 95% confidence limits from each of the three prediction models for each of the 14 surrogate taxa from which listed-species' LC_{50} values were derived are presented in Table 48. The effects estimates are presented in Table 49for the listed species (i.e., matches up the effects estimates for surrogate taxa in Table 48 with the listed species linked to each surrogate taxon).

The EC_{10} and EC_{20} concentrations for each of our three regression models were also estimated. The fathead minnow regression yielded an estimated EC_{10} of 4.4 µg/L CN (95% CI = 2.6-6.2 µg/L CN) and an estimated EC_{20} of 5.5 µg/L CN (95% CI = 3.5-7.4). By comparison, Gensemer et al. (2007) estimated an EC_{20} of 6.0 µg/L CN from a log-probit analysis of the fathead minnow data, but did not report confidence limits for that estimate. The brook trout regression yielded an estimated EC_{10} of 2.6 µg/L CN (95% CI = 0.0-8.4 µg/L CN) and an estimated EC_{20} of 4.1 µg/L CN (95% CI = 0.0-11.1). Gensemer et al. (2007) estimated an EC_{20} of 7.7 µg/L by linear interpolation of the brook trout data, and again did not report confidence limits for that estimate. The bluegill regression yielded an estimated EC_{10} of 4.6 µg/L CN (95% CI = 0.0-10.5 µg/L CN) and an estimated EC_{20} of 5.3 µg/L CN (95% CI = 0.0-11.5). Gensemer et al. (2007) estimated an EC_{20} of 5.6 µg/L CN from a log-probit analysis of the bluegill data, and also estimated an EC_{20} of 8.9 µg/L CN for the bluegill data from EPA's TRAP program. All of Gensemer et al.'s (2007) estimates fall within our 95% confidence limits, and in general show excellent agreement with our results even though Gensemer et al's methods differed from ours. This suggests that our results are not highly dependent on the particular statistical approach that we chose for our analysis.

Table 51. Estimated magnitude of effect of cyanide (at the CCC, 5.2 µg CN/L) on surrogate taxa for listed fish species (95% CL)*

Surrogate taxa used to estimate magnitude of effect on listed species	**Surrogate species**		
	Fathead Minnow	**Brook Trout**	**Bluegill**
	Reduction in the mean number of hatched eggs per spawn compared to controls	**Reduction in the mean number of viable eggs per spawn compared to controls**	**Reduction in the number of surviving larvae/juveniles compared to controls**
Actinopterygii (class**)**	48% (39%, 56%)	30% (1%, 55%)	56% (3%, 82%)
Order Cypriniformes	39% (28%, 49%	26% (0%, 54%)	44% (0%, 80%)

Family Catostomidae			
Xyrauchen texanus (species)	39% (28%, 49%)	26% (0%, 54%)	44% (0%, 80%)
Cyprinidae (family)	29% (15%, 42%)	21% (0%, 53%)	30% (0%, 78%)
Cyprinella monacha (species)	68% (63%, 72%)	42% (23%, 58%)	76% (50%, 89%)
Gila elegans (species)	57% (51%, 63%)	36% (12%, 56%)	66% (30%, 84%)
Notropis mekistocholas (species)	59% (53%, 65%)	37% (14%, 56%)	68% (34%, 85%)
Ptychocheilus lucius (species)	63% (57%, 68%)	39% (18%, 57%)	71% (41%, 86%)
Order Perciformes	36% (24%, 47%)	24% 0%, 53%)	40% 0%, 79%)
Percidae (family	63% (58%, 68%)	39% (18%, 57%)	72% (43%, 87%)
Etheostoma (genus	65% (60%, 70%)	40% (20%, 58%)	74% (46%, 88%)
Etheostoma fonticola (species)	81% (76%, 85%)	52% (37%, 64%)	86% (64%, 95%)
Order Salmoniformes, Family Salmonidae			
Oncorhynchus (genus)	60% (54%, 65%)	37% (15%, 57%)	69% (36%, 85)
Oncorhynchus apache (species)	87% (82%, 91%)	56% (42%, 68%)	90% (67%, 97%)
Oncorhynchus tschawytscha (species)	71% (67%, 76%)	45% (27%, 59%)	79% (55%, 91%)
Oncorhynchus kisutch (species)	71% (66%, 75%)	44% (27%, 59%)	79% (55%, 90%)
Oncorhynchus mykiss (species)	52% (45%, 59%)	33% (6%, 55%)	61% (16%, 83%)
Oncorhynchus clarki henshawi (species)	80% (75%, 84%)	51% (36%, 63%)	85% (63%, 94%)
Salvelinus (genus)	87% (83%, 92%)	57% (43%, 69%)	90% (68%, 97%)

*The magnitude of effect was estimated using the regression model for each surrogate response species and SS EC_X value for each surrogate taxa (Table 40). For each surrogate taxa there were two estimates of effects on reproductive performance and one estimate of effects on early life stage survival.

Table 52. Estimated magnitude of effect of cyanide (at the CCC, 5.2 µg CN/L) on listed fish species (95% CL). There are two estimates for effects on fecundity and one estimate for effects on early life stage survival for seven listed species due to exposure at based on surrogate species data.

Listed Species	Surrogate Taxa	Estimated reduction in fecundity and larvae/juvenile survival due to cyanide exposure at the CCC based on surrogate species data sets		
		Fathead minnow[1] (Percent reduction in the mean number of hatched eggs per spawn compared to controls)	Brook trout[2] (Percent reduction in the mean number of viable eggs per spawn compared to controls)	Bluegill[3] (Percent reduction in the number of surviving larvae/juveniles compared to controls)
Coho salmon *(Oncorhynchus kisutch)*	*Oncorhynchus kisutch*	71 (66, 75)	44 (27, 59)	79 (55, 90)
Chinook salmon (*Oncorhynchus tschawytscha)*	*Oncorhynchus tschawytscha*	71 (67, 76)	45 (27, 59)	79 (55, 91)
Chum salmon *(Oncorhynchus keta)*	*Oncorhynchus (genus)*	60 (54, 65)	37 (15, 57)	69 (36, 85)
Sockeye salmon *(Oncorhynchus nerka)*	*Oncorhynchus (genus)*	60 (54, 65)	37 (15, 57)	69 (36, 85)
Steelhead (*Oncorhynchus mykiss)*	*Oncorhynchus mykiss*	52 (45, 59)	33 (6, 55)	61 (16, 83)
Shortnose sturgeon *(Acipenser brevirostrum)*	Actinopterygii (class)	48 (39, 56)	30 (1, 55)	56 (3, 82)
Green sturgeon *(Acipenser medirostris)*	Actinopterygii (class)	48 (39, 56)	30 (1, 55)	56 (3, 82)

[1]Based on data contained in Lind et al. 1977
[2]Based on data contained in Koenst et al. 1977
[3]Based on data contained in Kimball et al. 1978
[4]Based on data contained in Schimmel 1981

Other Effects Estimates

The estimates of effects presented in Table 49 are based largely on ICE LCL (lower confidence limit) LC_{50} values for listed fish evaluation species. Those are the LC_{50} values that we accept as sufficiently accounting for the uncertainties inherent in relying on surrogate data and numerous other untested assumptions to estimate the sensitivity of listed species to cyanide. The Service, NMFS, and EPA agreed that using ICE LCL values was preferable to the practice of applying arbitrary uncertainty factors.

However, EPA has, at various times, questioned whether the use of ICE LCL values might not be overly conservative. Therefore, we also estimated effect levels using ICE MLE (maximum likelihood estimates) LC_{50} values for listed fish evaluation species (via revised $SSEC_x$ estimates). Those results are presented in Appendix F. Based on the fathead minnow prediction model, which was the strongest model, the median levels of effect predicted for the 15 ICE surrogate taxa were 51% and 65%, respectively, for ICE MLE and ICE LCL. The number of surrogate taxa with a predicted effect of 35% or greater was 11 and 14, respectively, for ICE MLE and ICE LCL. Those differences indicate only modest conservatism conferred by ICE LCL-based effects estimates as compared to ICE MLE-based estimates. Such modest differences would not have a decision-making impact. For both sets of results, unacceptably high levels of effect would overwhelmingly be the predominant prediction.

Empirical Test of Method Performance

Because only three concentration-response datasets are available, there is almost no basis for testing our method performance (i.e., there are no known directly measured "true" values for effects to our listed fish evaluation species at a concentration of 5.2 µg/L CN). However, because the fathead minnow and brook trout datasets focused on essentially the same response variable (number of hatchable/viable eggs produced per spawn) we can perform two tests of method performance. For each species, we can directly estimate a predicted effect level at 5.2 µg/L CN using the species-specific regressions. Those would be our estimates of the "true" effect level. Next, we can use our surrogate method and estimate an $SSEC_x$ for each species on the other species' response curve and evaluate the predicted effect level for that $SSEC_x$ value and compare the surrogate estimate to the estimated "true" value. The results are as follows:

> The directly estimated fathead minnow effect level at 5.2 µg/L CN is 18% with a 95% CI of 0%-34%. The fathead minnow $SSEC_x$ value on the brook trout response curve would be 3.2 µg/L CN, which yields an effects estimate of 15%. That is nearly identical to estimated "true" value and easily within the 95% CI for the "true value".

> The directly estimated brook trout effect level at 5.2 µg/L CN is 25% with a 95% CI of 0%-54%. The brook trout $SSEC_x$ value on the fathead minnow response curve would be 8.4 µg/L CN, which yields an effects estimate of 38%. Again, that is within the 95% CI for the "true" value, although our estimate of the "true" value is not very precise and therefore the 95% CI is fairly wide.

In summary, in both test cases, the estimated effect level derived from our surrogate methodology is not significantly different from the estimated "true" value in a statistical sense, but the second comparison has low statistical power. Further validation testing of this sort should be done as concentration-response datasets become available for more species using a comparable response variable, but it is reassuring that in these test cases our method yielded results that were nearly identical to the "true" value in one case and reasonably close to the "true" value in the other case.

Derivation of the Criterion Continuous Concentration (CCC)

Our analysis predicts that the listed fish species considered in this Opinion would be severely affected by exposure to cyanide at the CCC. National criteria are typically derived using chronic toxicity data from laboratory tests. As noted earlier, most aquatic life criteria that have been derived thus far, including the cyanide criterion, chronic values have been obtained by calculating the geometric mean of the lower and upper chronic limits. In practice, the upper and lower chronic limits are often statistically determined by hypothesis testing. The lower limit is typically the NOEC, which is defined as the highest test concentration where the effects are not statistically significantly different from controls. The upper limit is typically the LOEC, which is defined as the lowest test concentration where the effects are statistically significantly different from controls. The guidelines recommend that the magnitude of effect associated with the upper and lower chronic limits should be considered when determining values that appropriately estimate acceptable and unacceptable levels of adverse effect:

> *Because various authors have used a variety of terms and definitions to interpret and report results from chronic tests, reported results should be reviewed carefully. The amount of effect that is considered unacceptable is often based on a statistical hypothesis test, but might also be defined in terms of a specified percent reduction from the controls. A small percent reduction (e.g., 3%) might be considered acceptable even if it is statistically significantly different from the control, whereas, a large percent reduction (e.g., 30%) might be considered unacceptable even if it is not statistically significant."*

Based on this guidance, the threshold for unacceptable adverse effects would be estimated by the chronic value. The magnitude of effect at the threshold would then be equivalent to the magnitude of effect at the chronic value. For chronic criteria derived using hypothesis tests, this would be the magnitude of effect occurring at a concentration equal to the geometric mean of the NOEC and LOEC, that is, somewhere between an acceptable and unacceptable level of adverse effect. The guidelines do not specify a level of adverse effect on which the threshold for unacceptability should be based. The only mention of a numeric value or range is provided in the guidance for selecting chronic limits (mentioned above) and suggests that this threshold may lie between 3% and 30%.

Thus, for a given species or test the magnitude of effect at the chronic value will depend on the magnitude of effect at the lower and upper chronic limits. We followed this approach for estimating the magnitude of effect occurring at the cyanide CCC. The freshwater cyanide CCC was derived based on chronic toxicity data for 4 species (Table 50): 3 fish (fathead minnow, brook trout, and bluegill) and 1 invertebrate (*Gammarus pseudolimnaeus*). Chronic values for each species were obtained by calculating the geometric mean of the lower and upper chronic limits. The magnitude of effect at the lower and upper chronic limits was calculated by comparing responses at the lower and upper limits to controls. For fathead minnow and brook trout these effects were expressed as reduction in the mean number of eggs spawned per female compared to controls; for the bluegill the effect was reduction in larvae/juvenile survival compared to controls; and for *G. pseudolimnaeus* the effect was a reduction in the mean number of eggs or young per gravid female relative to controls.

We then estimated the magnitude of effect at the chronic value by linear interpolation between lower and upper chronic limits (Table 50). Based on these calculations the magnitude of effect at the chronic values for the fathead minnow, brook trout, bluegill and *G. pseudolimnaeus* would be 52%, 32%, 54%, and 47%, respectively. According to the guidelines, if there were a sufficient number of chronic values (i.e., chronic values for species in 8 phylogenetic families) the chronic criterion could be computed directly from the distribution of chronic values (see earlier discussion under *Derivation of Criteria*). If there were fewer chronic values, as was the case for cyanide, the chronic criterion would be computed using ACRs. ACRs for the 4 freshwater species were reported in the cyanide criterion document and are shown in Table 50. The ACRs were calculated by dividing the species mean acute value (i.e., mean LC_{50} for the species) by the chronic value. For example, the ACR for fathead minnow (7.633) was computed by dividing 125.1 µg CN/L (the mean LC_{50} for the species) by 16.39 µg/ CN/L (the chronic value). Thus, the ACR is the ratio between the concentration of cyanide causing 50% lethality (following acute exposure) and the concentration following chronic exposure that causes a level of adverse effect that is at the threshold of unacceptability, i.e., 52% for fathead minnow. The guidelines require that, for criteria derivation, the geometric mean of individual species ACRs is used to obtain the Final ACR. For cyanide, the freshwater Final ACR was 8.562 (Table 50). We estimated the magnitude of chronic effects associated with the Final ACR to be about 45% (Table 50).

The Final ACR and the FAV were then used to derive the CCC. The guidelines describe how the FAV is computed. In short, the FAV is set equal to the 5th percentile estimate from the distribution of genus mean acute values. In other words, the FAV represents the genus with acute sensitivity (LC_{50}) in the *sensitive tail* of the distribution where, theoretically, approximately 5% of the genera would be more sensitive and about 95% of the genera would be less sensitive. Based on this analysis, the FAV for cyanide was determined to be 62.68 µg CN /L. Since the guidelines include provisions for adjusting the FAV to protect commercially and recreationally important species, EPA lowered the FAV from 62.68 µg/L to 44.73 µg/L because the SMAV for rainbow trout (44.73 µg/L) was below the calculated FAV. The cyanide criterion (5.2 µg/L) was then derived by division of the FAV (44.73 µg/L) by the Final ACR (8.562). Thus the chronic criterion, 5.2 µg CN/L, was based on the concentration intended to protect rainbow trout from unacceptable adverse effects. Based on our estimate of the magnitude of effect associated with the Final ACR, we estimate the magnitude of adverse effects occurring to rainbow trout at the chronic criterion to be approximately 45%. This value is higher than we would have expected considering it is intended to represent the threshold for unacceptable adverse effects. However, the magnitude is in line with effects we predicted for the other listed fish species, most of which were estimated to be as (or more) sensitive to cyanide as rainbow trout.

Table 53. Chronic toxicity data used by EPA to derive the freshwater chronic criterion for cyanide. **Effect levels were calculated using data from the original papers.**

Species	Chronic Limits[1]				Chronic Value[2]		LC_{50}[3] (µg CN/L)	ACR[3]
	Lower		Upper					
	(µg CN/L)	Effect	(µg CN/L)	Effect	(µg CN/L)	Effect		
Fathead Minnow	13.3	47%	20.2	58%	16.39	52%	125.1	7.633
Brook Trout	5.6	18%	11.0	53%	7.849	32%	83.14	10.59
Bluegill	9.3	30%	19.8	88%	13.57	54%	99.28	7.3
Gammarus	16	0%	21	100%	18.33	47%	167	9.111
Geometric mean						45%		8.562

[1] Lower and upper chronic limits were taken from the cyanide criteria document. For fathead minnow and bluegill these values were determined statistically (i.e., NOEC and LOEC identified via hypothesis tests). Effect levels were take from Tables 5, 8 and 10 in the Effects section of the BO and from Oseid and Smith 1979.
[2] Chronic values were taken from the cyanide criteria document. Effect levels associated with the chronic values were estimated by linear interpolation between the effects at the lower and upper chronic limits.
[3] Acute-Chronic Ratios were taken from the cyanide criteria document.

This same conclusion, that NOEC/LOEC-based estimates of "chronic values" can correspond to ≥ 40% adverse effect, has also been reached by others. Decades ago Suter et al. (1987) reported that MATC's for fish fecundity, on average, corresponded to a 42% level of adverse effect (MATC = Maximum Acceptable Toxicant Concentration; a term for the geometric mean of the NOEC and LOEC from a given toxicity test and often assigned by EPA as the estimated "chronic value" from a test). Other response endpoints were found to correspond to average adverse effect levels of 12-35%. More recently, SETAC (Society for Environmental Toxicology and Chemistry) convened a panel of experts (Reiley et al. 2003) who concluded that "*...[toxicity] tests with high variability may result in an(sic) NOEC corresponding to a response greater than 40% different from the control.*" Moore and Caux (1997) statistically examined nearly 200 toxicity data sets and found that most NOEC's (76.9%) exceeded a 10% adverse effect level and most LOEC's (62.4%) exceeded a 30% effect level. Various other researchers have noted a variety of adverse effect levels for NOEC's, such that Crane and Newman (2000) were led in summary to conclude that "*… [NOEC] effect levels from individual tests ranged from nearly 0% to nearly 100%.*" For seven cyanide toxicity tests with sufficient data for comparison, Gensemer et al. (2007: Figure 3-7) found in all cases that the geometric mean of the NOEC and LOEC corresponded to an adverse effect level of ≥ 20% (how much greater was not reported).

Because of the highly variable and often substantive levels of effect associated with NOEC's, LOEC's, MATC's, and with the "chronic values" based on them, and for numerous other reasons, a strong professional consensus recommendation to avoid using NOEC/LOEC-based estimates for regulatory thresholds (when possible) has been expressed repeatedly. For example, there was an ISO (International Organization for Standardization) resolution (ISO TC147/SC5/WG10 Antalya 3) as well as an OECD (Organisation for Economic Co-operation and Development) workshop recommendation (OECD 1998) that the NOEC should be phased out from international standards (OECD 2006:14). Environment Canada (2005) notes, that there is a growing literature which points out many deficiencies of the NOEC approach (Andersen et al. 2000; Bailer and Oris 1999; Chapman 1996; Chapman et al. 1996; Crane and Godolphin 2000; Crane and Newman 2000; Miller et al. 1993; Moore and Caux 1997; Noppert et al. 1994; Pack 1993; Pack 1998; Suter et al. 1987; Suter 1996). Moving away from the NOEC/LOEC approach was also among the recommendations of the SETAC panel for improving the scientific basis of water-quality criteria (Reiley et al. 2003).

Accordingly, EPA has begun employing a regression approach for estimating "chronic values" whenever sufficient data are available to do so. For example, in the 1999 update for ammonia water quality criteria EPA used regression analyses to estimate 20% effect concentrations (EC_{20}s) from individual toxicity tests and used those EC_{20}s as estimates of chronic values (EPA 1999). Likewise, estimated EC_{20}s have been the basis for estimating chronic values in recently proposed updates for copper and selenium water quality criteria (EPA 2003a, 2004). EPA's choice of the EC_{20} as a basis for estimating chronic values was justified from statistical considerations rather than from biological or demographic considerations:

> *To make [chronic values] reflect a uniform level of effect, regression analysis was used here both to demonstrate that a significant concentration-effect relationship was present and to estimate [chronic values] with a consistent level of effect. Use of regression analysis is provided for on page 39 of the 1985 Guidelines (Stephan et al. 1985). The most precise estimates of effect concentrations can generally be made for 50 percent reduction (EC50); however, such a major reduction is not necessarily consistent with criteria providing adequate protection. In contrast, a concentration that caused a low level of reduction, such as an EC5 or EC10, is rarely statistically significantly different from the control treatment. As a compromise, the EC20 is used here as representing a low level of effect that is generally significantly different from the control treatment across the useful chronic datasets that are available for ammonia.*

Pack (1993) asserted that most ecotoxicologists consider effects in the range of 5-20% to be biologically acceptable depending on the species involved and the type of effect. However, EPA appears to have chosen the top end of that range based more on the expected statistical power of toxicity tests than on a serious examination of the typical demographic sensitivity of biotic populations to a 20% adverse effect on survival, growth, or reproduction. Furthermore, 95% statistical confidence limits for most EC_{20} estimates are likely to extend well into adverse effect levels that would be of unquestionably serious demographic concern for most organisms. As evident from the above discussion, most chronic criteria derived by EPA, including for cyanide, are highly likely to be associated with $\geq$ 20% adverse effect level for species at the vulnerable end of species sensitivity distributions (such as the subset of ESA-listed species we are evaluating). Therefore, it should be no surprise that our estimated effect levels for such species at the current cyanide CCC of 5.2 µg/L are almost always higher than 20% and in some cases substantially higher.

Population Responses to Reductions in Fecundity and Juvenile Survival

Laboratory experiments have demonstrated that even closely related fish species can demonstrate great differences in sensitivity when exposed to the same chemical, as measured by differences in acute or chronic toxicity values. This variability in sensitivity has been related to differences in species' physiology and life history strategies. Similarly, population modeling and experimental studies have shown that variation in population-level responses to environmental toxicity can also be expected among species as a consequence of factors such as life history strategies, life stage affected, and density dependence. Studies have also demonstrated that chronic toxicity can lead to population decline and extirpation.

Under the ESA, in determining whether a proposed Federal action is likely to jeopardize the continued existence of a listed species under the ESA, we assess whether the proposed activity reasonably would be expected to appreciably reduce the likelihood of survival and recovery of a listed species by reducing its reproduction, numbers, or distribution. Two common metrics used in population modeling to assess effects of perturbations on populations are population growth rate and time to or probability of extinction.

Population growth rate is the change in a population size over a unit time period. Long-term reductions in population growth rate as low as 5% has been shown to significantly increase a population's likelihood of extinction (Snell and Serra 2000). Population growth rate can be positive when the population is increasing, negative when decreasing, or zero when the net difference between births, deaths, and migration is zero and the population is stable. For listed species, populations may exist in any of these states depending on its recovery status. Our analysis determines the relative predicted effects of the action to the population growth rate, regardless of its starting value.

Using known parameters of a species' life history, sensitivity analyses can be conducted to determine which parameters, when modified, will have the greatest impact on the species' population growth rate. Elasticity analysis is one type of sensitivity analysis that is commonly used in conservation biology to demonstrate the relative contributions to population growth rate made by life cycle transitions, based on vital rate statistics for survival, growth and fertility. While these types of analyses cannot predict absolute effects to population size, because they quantify the relative importance of an element to changes in population growth rate, they can help focus management decisions on those demographic parameters that exhibit the largest elasticity, and thus, the largest impact on population growth (de Kroon et al. 2000). However, elasticity analysis requires the development of a population model, for which adequate data are often scarce. Because this type of demographic data is often lacking for threatened and endangered species in particular, the need to develop generalized approaches for classifying population responses to perturbation for rare species has been recognized (Dennis et al. 1991; Heppell et al. 2000).

Several authors have examined the effect of life history strategies on the elasticities of various demographic measures. In evaluating demographic parameters of 50 mammal populations with different life history strategies, Heppell et al. (2000) found that phylogeny alone is often not a reliable indicator of which vital rates (survival, growth and fertility) will have the greatest impact on elasticity. Instead, the authors found that species that mature early and have high reproductive output had high fertility elasticities and low adult survival elasticities. Conversely, for those which mature late and have long lifespans, fecundity and early offspring survival are less important than survival of juveniles to maturity to changes in population growth rate. Calow et al. (1997) also found that the relative importance of juvenile fish survival can vary according to reproductive strategy. These authors concluded that reductions in juvenile survival would have the greatest impact on semelparous fish species, in which adults die after reproduction, a lesser impact on a moderately iteroparous population, in which adult postreproductive survival is intermediate, and the least impact on strongly iteroparous species, in which adult survival after reproduction is high. These assumptions held true for elasticity analysis of the green sturgeon, a fish species with life history patterns such as late-maturity and long-life that are common to other sturgeon (Heppell 2007).

Juvenile survival had relatively lower elasticity values than adult and subadult survival, with compensation for the loss of adults requiring much larger increases in young-of-the-year survival than would be commensurate with the loss. However, other authors have found increased importance of juvenile survival for sturgeon, despite their lifespan (Gross et al. 2002; Paragamian and Hansen 2008). Gross et al. (2002) hypothesized that this difference was due to the vastly larger fecundity of sturgeon as compared to other long-lived species.

Vélez-Espino et al. (2006) argue the need for a broadscale summary of species' population dynamics to help guide the conservation biology of freshwater fishes, for which information on life history is often limited. Using information, on adult survival, juvenile survival, and fecundity, the authors performed elasticity analyses on 88 species of freshwater fish and found that they could be classified into 4 functional groups with regard to the sensitivity of their population growth rates:

1. species most sensitive to perturbations in adult survival
2. species most sensitive to perturbations to adult and juvenile survival
3. species most sensitive to perturbations to juvenile survival
4. species most sensitive to perturbations to juvenile survival and fecundity

These groups are characterized by decreased age at maturity, longevity, and reproductive lifespan as one moves from group 1 to group 4. Age at maturity, reproductive lifespan, fecundity, juvenile survivorship, and longevity were all correlated with adult survival and fecundity. However, the best predictors of elasticity patterns were longevity, which explained 93% of the variability in the elasticity of adult survival, and age at maturity, which explained 92% of the variability in the elasticity of fecundity. The authors also found that elasticities are highly conserved among genera within the same taxonomic family

Spromberg and Birge (2005) also found that life history strategies influence effects to populations. The five life history strategies they modeled encompassed differences in stage-specific survival, fecundity and hatch success, number of spawning events, and life-span. The authors found that regardless of strategy, changes in the number of young-of-the-year stage individuals had the greatest impact on population growth rate. However, the relative contribution of this parameter was greatest for life history strategies with multiple spawnings, high fecundity, and short lifespans as opposed to those with longer lifespan, which had increased elasticity of adult survival.

Spromberg and Meador (2005) linked toxicant effects on immune suppression, reproductive development, and growth reduction to demographic traits in Chinook salmon and modeled their influence on population growth rate. Overall, effects to first- and second-year survival had the greatest elasticities, with constant reductions to first year survival as low as 10% achieving population declines ranging from 35-78% compared to controls. Other studies have demonstrated the importance of first year survival in this species (Kareiva et al. 2000). Spromberg and Meador (2005) also found that models which incorporated effects to both survival and reproduction were additive, indicating the importance of evaluating the overall impact of all potential impacts to population growth.

Many listed species populations are limited by the amount of adequate habitat or resources and experience some degree of density dependence. Density-dependence at any life stage must be considered in elasticity analysis in order to yield appropriate results (Grant and Benton 2000; Hayashi et al. 2008). In a review of toxicant impacts on density-limited populations, Forbes et al. (2001) noted that the full range of interactions have been found between toxicant stress and density dependence, including less than additive, additive, and more than additive effects. Also, the type of effect may vary with increasing toxicant concentration from one that ameliorates density dependent effects at low toxicant concentrations to one that exacerbates density dependent effects at higher toxicant concentrations. Case studies which incorporate density-dependence into population modeling demonstrate this variability, with overall impacts to populations shown to be both lesser (Van Kirk and Hill 2007) and greater (Hayashi et al. 2008) than the level of effect that would be predicted from individual response depending on the situation. In time, density-dependant populations may rebound, stabilize at a lower absolute population number, or continue to decline until the population is extirpated (Forbes et al. 2001). Modeling exercises have demonstrated cases in which populations stabilize at new, lower equilibrium abundances in response to a constant impact (van Kirk and Hill 2007; Spromberg and Meador 2005).

A species' likelihood of persistence can also be estimated a number of ways. There are no standard methods or protocols to estimate the risk of extinction. Instead, the method used is usually dependent on the availability of data available on the species in question and species' biology. Extinction risk analyses methodologies may be qualitative, semi-quantitative, or quantitative. One quantitative method that is used widely for modeling a species' time to extinction or probability of extinction is Population viability analysis (PVA). PVAs use simulation modeling to identify threats to species and to assess the vulnerability of populations to extinction risks. These models incorporate demographic parameters such as fecundity, survivorship, age structure, and population size, but can also incorporate effects to the environment such as habitat degradation and catastrophic events. As for the evaluation of population growth rate, sensitivity analysis is used to determine which factors have the greatest impact on population persistence, and many experts feel that parsing out these influential factors for management purposes is the best utilization of these models, as opposed to absolute predictions of population decline. However, PVA models require a depth of demographic data that is often lacking for listed species.

For Pacific salmon, NMFS has not found a PVA that completely represents the various risks facing salmon populations (McElhany et al. 2000). Consequently NMFS created the viable salmonid population concept to provide useful benchmarks for evaluating actions that directly affect natural populations and for which incremental increases in extinction risk may be difficult or impossible to accurately quantify. Where PVAs have been conducted for specific populations of salmon, these have informed NMFS in status assessments. While the VSP concept isn't meant to replace quantitative models where they can be properly used because the VPS employs a combination of quantitative and qualitative methods for determining the extinction risk of listed species it is more flexible and easier to use where data are limited (McElhany et al. 2000). Under the VSP approach, risk is first addressed at the population level and then the ESU. Individual populations are assessed according to four parameters: abundance, growth rate/productivity, spatial structure, and diversity. NMFS focuses on these parameters because they are reasonable indicators of extinction risk (viability). Although, there is no formal link between VSP and jeopardy under Section 7, the same population level parameters used in VSP, whenever available, are a significant part of our analysis in determining whether an agency's action is likely to jeopardize the continued existence of a listed species.

Summary of Population Responses to Reductions in Fecundity and Juvenile Survival

Modeling and experimental studies have shown that chronic toxicity to pollutants can lead to population decline and extirpation. Variation in population-level responses to environmental toxicity can be expected among species as a consequence of factors like species life history strategies, life stage affected, density dependence, and magnitude of toxicant stress. Although the degree varied among different life history strategies, fecundity and juvenile survival remained a highly influential demographic parameter throughout modeled scenarios, with adult survival taking on greater importance in long-lived species. These results must be coupled with other influences on the population status, such as the degree of density dependence and additional environmental perturbations such as catastrophes. Although population modeling often requires more demographic information than is available for threatened and endangered species, careful selection of surrogates and use of their data may allow for extrapolation from models for species with similar life histories.

Summary of the Direct Effects

According to our analysis, Chinook, chum, coho, and sockeye salmon, and green and shortnose sturgeon exposed to cyanide are likely to experience reduced survival, reproduction, and may also experience effects on growth, swimming performance, condition, and development, as described above. Our analysis demonstrates that acute and chronic toxicity may be exacerbated by other stressors such as dissolved oxygen concentration, temperature, and the presence of other pollutants in the water column. That is, the threshold of adverse effects is diminished in the very cold waters and low dissolved oxygen conditions.

Relatively few studies were available for estimating the magnitude of effects that could occur following exposure to cyanide at criterion concentrations. Because no data for cyanide toxicity to sturgeon exist, LC50 values for sturgeon were derived from the 5% SSD concentration for the class Actinoptergyii, which encompasses all known cyanide toxicity data for fish. From this data, we developed quantitative estimates of the effects on fecundity, hatching success, and survival of young first-year fish (Table 49). Given the limited data set, our estimates are the same for green sturgeon and shortnose sturgeon, as well as sockeye salmon and chum salmon (the latter are based on data from the genus *Oncorhynchus*). Based on our analysis, we estimate that green and shortnose sturgeon exposed to cyanide at the CCC may experience a reduction in juvenile survival that is as high as, but not likely to be greater than, 56%. Our estimates reveal that the green and shortnose sturgeon may experience a reduction in the number of hatched eggs and that reduction could be as high as, but is not likely to be greater than, 30%.

Similarly, we expect that coho and Chinook salmon would experience a reduction of juvenile survival and that reduction could be as much as, but is not likely to be greater than 79%. We estimate that, when exposed to cyanide at the CCC, coho and Chinook salmon may experience a reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 45%. Similarly we expect that chum and sockeye salmon would experience a reduction in juvenile survival and that reduction could be as much as, but is not likely to be greater than 69%. We estimate that, when exposed to cyanide at the CCC, chum and sockeye salmon may experience a reduction in the number of hatched eggs and that reduction could be as much as, but is not likely to be greater than, 37%. Our estimates reveal that steelhead would experience a reduction in juvenile survival and that reduction could be as much as, but is not likely to be greater than 61%. We estimate that, when exposed to cyanide at the CCC, steelhead may experience a reduction in the number of hatched eggs and that reduction could be as high as, but is not likely to be greater than, 33%.

Young of the year fish, and juvenile fish that do survive exposure to cyanide could experience reduced growth rates which would increase their vulnerability to a host of potential stressors, including temperature, flow, and inter- and intraspecific competition for food and cover. We expect that such exposure could also delay reproductive maturity and productivity. These reductions in reproductive performance and survival represent reductions in the fitness of the individuals exposed to cyanide at the chronic criterion concentration. Changes in the fitness of the individuals in a population will affect the population as a whole, and could be measured in terms of changes in population growth rates and changes in risk of extinction.

Sturgeon have naturally high adult survival, and the loss of juvenile life stages is particularly problematic. Several authors have suggested that the rate of survival may be so high that management at the levels of these age classes is unlikely to improve their survival or increase population growth rate (Gross et al 2002; Heppell 2007). As such, recovery efforts are often based upon increasing survival in juvenile age classes. Gross et al (2002) modeled population growth rates for three species of sturgeon that varied in life history traits such as size, lifespan, age to maturity, and migration. All three sturgeons showed similar elasticity profiles, and thus the authors concluded that general interpretation could be applied to sturgeon across species. In contrast to other elasticity profiles for long-lived species, elasticity in sturgeon was highest in individual young-of-the-year and juvenile age classes, dropped at the onset of maturity, and continued to decline for each successive adult age class. Fecundity had relatively low elasticity, as the effects of changes in fecundity are shared among all adult age classes of these long-lived species, and the value of changes to egg numbers is lessened by the high mortality of the young-of-the-year age class. The authors concluded that population growth rate would show little response to improvements in fecundity, but greater responses in survival at either the young-of-the-year or juvenile age classes. However, since survival of the juvenile and adult age classes is naturally high, improvements at these stages will have smaller effects to improving population growth rate than increases to survival of young-of-the-year, when natural mortality is greater. The authors note that among biologists and managers involved in sturgeon conservation, habitat improvement was regarded as the most important conservation undertaking for sturgeon. Results from this study indicate that restoration efforts should target the survival of age classes with high elasticity, specifically young-of-year and juvenile. Paragamian and Hansen (2008) drew similar conclusions in modeling effects on population growth of the Kootenai River white sturgeon. The authors found that subadult and adult survival (>90%) was much higher than that of juveniles (40% in the first year), and recovery was dependent on increasing first-year survival. The authors suggested that to have the largest effect on recovery, managers should increase the current targeted recruitment rate.

Unlike sturgeon, most Pacific salmon (with the exception of steelhead and cutthroat trout) are semelparous, such that they spawn only once. Consequently, reductions in the number of viable eggs and juvenile survival through their first year would likely have greater population-level effects on Chinook, coho, sockeye, and chum salmon. Low fresh water is survival is considered typical of most salmon populations, although estimates for many populations are nonexistent, mortality rates are recorded from 40-90% (Sandercock 1991; Bradford 1997). According to Brandford, the coefficient of variation (CV) for interannual survival in fresh water is about 30% averaged over all species. The factors that influence the freshwater survival rate for the likely differs somewhat between widely-dispersed spawning species (e.g., steelhead, coho and Chinook salmon) compared to those that spawn in dense aggregations (e.g., sockeye and chum salmon), as well as the length of time spent in freshwater rearing (e.g., coho salmon versus early migrant Chinook salmon or chum salmon). For Pacific salmon, mortality appears to be roughly equally divided between fresh water and marine waters, suggesting that each habitat contributes to recruitment variation (Bradford 1997). Consequently, significant reductions in freshwater production would be expected to significantly affect the number of adults returning to fresh water to spawn.

As discussed earlier, there are several factors that can influence the relative toxicity of chemical contaminants under natural exposure conditions. When organisms are stressed due to environmental factors outside their normal optima they may become more sensitive to a given toxicant. This can occur when homeostasis is disrupted in organisms that are infected with a pathogen, outside their normal range for various water quality parameters (salinity, pH, or temperature), diseased, or debilitated due to other toxic insults. Very cold temperatures and low DO conditions increase the toxicity of cyanide. Despite the limited number of studies on these influencing factors, until more work can be done we have little evidence to suggest species specific responses to cyanide under low DO conditions or low water temperatures. Considering that cyanide is a respiratory toxin that inhibits oxidative metabolism, it is not surprising that the effects are exacerbated under conditions where oxygen availability is limited. Any factor that affects gill ventilation will also likely affect the amount and speed at which the toxin is distributed in the body. A fish is under stressed conditions like oxygen depletion, would typically increase their ventilation rate to compensate for the low DO and would, in this situation also increase their rate of uptake of aqueous cyanide.

In summary, exposure to aqueous cyanide at the approved CCC and CMC is likely to lead to the fitness consequences for Chinook, coho, sockeye, and chum salmon, and green and shortnose sturgeon. In particular, exposure to cyanide concentrations at the chronic criterion could substantially reduce reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and reducing the survival of young fish through the first year.

Sturgeon and salmon may also experience effects on growth, swimming performance, condition, and development. While sturgeon have developed a life history that allows them to cope with low survivorship to maturity and occasional decreases in recruitment, these adaptations are unlikely to compensate for a constant reduction in both fecundity and early life stage survival. The reductions we estimate in survival of young fish through the first year in particular would substantially decrease survival and recovery of this species. Because of the high magnitude of effects, we would expect density-dependent compensatory mechanisms, if they exist, to be overwhelmed.

Based on our analysis, we expect that the proposed action would significantly reduce the absolute numbers of green sturgeon, shortnose sturgeon, Chinook salmon, chum salmon, sockeye salmon, coho salmon, and steelhead. Based upon the magnitude of effects we anticipate could occur, the distributions of green and shortnose sturgeons are likely to be reduced in waters where they are exposed to cyanide at the levels defined by the chronic criterion, and may be reduced when cyanide exposure overlaps with low water temperatures or low DO concentrations.

Critical Habitat

We evaluated the effect of EPA’s approval of the cyanide water quality standards on the effect of critical habitat by first reviewing the essential features or primary constituent elements of critical habitat for listed and proposed designations. Based on our analysis, the primary features that may be affected by EPA’s approved water quality criteria are those designated as “water quality” areas for growth, development and reproduction (salmon and green sturgeon). We evaluated the “water quality” feature according to whether the acute or chronic criteria were likely to reduce the amount of clean water available for supporting essential patterns of growth, development or reproduction.

Approval of the CCC in state water quality standards would allow states to manage cyanide in waters to these levels. Even if waters never systematically reached these levels, the use of the aquatic life criteria in NPDES permits, TMDL limits, indicates the importance that these numeric values play in the overall success and operation of the water quality program. Our analysis demonstrates that where cyanide concentrations reach the approved standard, the proposed action would likely adversely affect the quality of water to the degree that it would impair individual reproduction and survival of green sturgeon, Chinook salmon, coho salmon, chum salmon, sockeye salmon and steelhead, and would cause these species to experience adverse effects to growth, swimming performance, condition, and development. For green sturgeon, we estimate the reduction in the number of hatched eggs could be as high 48% and the reduction in the survival of young fish through the first year as high as 56%. For coho and Chinook salmon, we estimate the reduction in the number of hatched eggs could be as high 45% and the reduction in the survival of young fish through the first year as high as 79%. For chum and sockeye salmon, we estimate the reduction in the number of hatched eggs could be as high 37% and the reduction in the survival of young fish through the first year as high as 69%. For steelhead, we estimate the reduction in the number of hatched eggs could be as high 33% and the reduction in the survival of young fish through the first year as high as 61%. These effects are estimated to be of a magnitude great enough to reduce numbers of both sturgeon and salmon. Approval of the CCC would adversely affect the quality of water to the degree that normal population growth would be severely reduced, and sturgeon and salmon may be extirpated from critical habitat containing cyanide at approved values. Not only would impacts to water quality resulting from management of cyanide to the CCC diminish the ability of critical habitat to provide for conservation of the these species, our analysis also suggests that the conservation value of critical habitat for these species would likely be diminished at concentrations below EPA's recommended CCC for fresh water.

The Impacts of Reduced Salmon Populations – Summary of Indirect Effects

Salmon are a significant contributor to the overall ecological food web throughout their range, whether they are from listed populations or unlisted populations. Two significant indirect effects of the proposed action, attributable not to the direct toxicity of cyanide, but the action's impact on Chinook, coho, sockeye and chum salmon and steelhead population abundance would include the further loss of primary prey species for southern resident killer whales and Cook Inlet beluga whales, and the loss of salmon nutrient transport to freshwater systems, which indirectly affects their own productivity. Bilby et al. (1996) demonstrate that juvenile and older age classes of salmon grow more rapidly with the appearance of spawners because these younger fish will feed on eggs and spawner carcasses. Salmon carcasses strewn along river reaches and streambanks are a significant source of food to a wide number of animals and affect the overall productivity of nutrient-poor systems (Bilby et al. 1996; Cederholm et al. 2000). The loss of these "marine derived nutrients" likely significantly reduces the survival of their own species, particularly in nutrient poor streams. Bilby et al. (1996) demonstrated that the mean fork length of juveniles and up to 45% of the carbon in cutthroat trout and 40% of the carbon in young of the year coho comes from the decaying carcasses of the previous generation of salmon. The increased body size is directly correlated to increases in over winter survival and marine survival. Based on historical cannery records and current records of escapement, Gresh et al. (2000) estimate this nutrient source has declined to about 13 to 17 percent of the historic biomass of return salmon to Pacific Northwest streams (Washington, Oregon, Idaho, and California). They suggest that this loss is one important indicator of ecosystem failure, contributing to the observed reductions in abundance we have seen in many salmon populations, and could further hamper recovery efforts. Thus, while we may have estimated the direct loss of individuals attributable to the proposed action, further reductions in many populations would be expected as adult spawner numbers decline from reduced recruitment attributable to the proposed action.

Similarly, although not obligate feeders, southern resident killer whales feed primarily on salmon and salmon are seasonally an important prey for Cook Inlet beluga whales. The reductions in salmon populations anticipated as a result of this action can be expected to have significant affects on southern resident killer whales and their critical habitat, and Cook Inlet beluga whales and their proposed critical habitat. Based on killer whale stomach contents from stranded whales and field observations of predation, Ford et al. (1998) determined that 95% of the diet of resident killer whales consists of fish, with a significant portion being Chinook salmon (about 2/3 of the samples that were identified to species). The authors suggested that Chinook salmon may be preferentially hunted by killer whales because of their large body size, high fat content, and seasonal distribution patterns. Although, Cook Inlet beluga whales feed on a variety of other fish species Pacific salmon are an important prey species for these animals as they build their lipid body stores essential to their winter survival. The significant reduction in the southern resident killer whale's primary prey species, Pacific salmon in general and in particular Chinook salmon, from the proposed action is likely to have significant effects on the fitness of southern resident killer whales and their population viability. As noted earlier, a 50% reduction in killer whale calving has been correlated with years of low Chinook salmon abundance (Ward et al. 2009a). Cook Inlet beluga whales would similarly experience a significant reduction in their most abundant summer and fall prey species (most of which, are non-listed Chinook, coho, sockeye, and chum species, although some listed species may be consumed during their marine migrations to Alaska). The proposed action, based on our analysis would significantly reduce freshwater production of all listed salmon species, as well as non-listed salmon species where cyanide concentrations are allowed to reach EPA's recommended aquatic life criteria concentrations. As noted earlier, we expect the proposed action would cause as high as a 79% reduction in the survival of juvenile (young fish through their first year) Chinook salmon, and as high as a 45% reduction in the number of viable eggs. These losses would severely reduce the number of adult Chinook salmon in the Puget Sound ESU, and would reduce the forage base for southern resident killer whales. Southern resident killer whales are not restricted to Puget Sound, but do spend a large portion of time in Puget Sound, the Strait of Juan de Fuca, and Haro Strait. Prey losses would also be realized throughout their range, including Oregon and California. Consequently, we expect that the proposed action would significantly reduce the absolute numbers of southern resident killer whales by reducing the absolute numbers of their primary prey. Based upon the magnitude of effects estimated to salmon, we expect the numbers, distribution and reproduction of southern killer whales would likely to be reduced due to significantly a reduced forage base.

Similarly, we expect the proposed action would cause as high as a 79% reduction in the survival of juvenile (young fish through their first year) coho salmon, as high as a 69% reduction in the survival of juvenile sockeye and chum salmon, and as high as a 44% reduction in the number of viable coho salmon eggs, and as high as a 37% reduction in the number of viable sockeye and chum salmon eggs. These losses would severely reduce the forage base of Cook Inlet beluga whales, and as a result we expect that the proposed action would significantly reduce the absolute numbers of Cook Inlet beluga whales by reducing important prey species. Based upon the magnitude of effects estimated to salmon, we expect the numbers, distribution and reproduction of Cook Inlet beluga whales would likely be reduced due to a significantly a reduced forage base.

Critical Habitat of Southern Resident Killer Whales

We evaluated the effect of EPA's approval of the cyanide water quality standards on the effect of critical habitat by first reviewing the essential features or primary constituent elements of critical habitat for listed designations. Based on our analysis, the primary features that may be affected by EPA's approved water quality criteria are those designated as "prey species of sufficient quantity, quality, and availability to support individual growth, reproduction and development, as well as overall population growth." Based on our analysis, we estimate that coho and Chinook salmon will experience reductions in the number of hatched eggs as high 45% and the reduction in the survival of young fish through the first year as high as 79%. For chum and sockeye salmon, we estimate the reduction in the number of hatched eggs could be as high 37% and the reduction in the survival of young fish through the first year as high as 69%. For steelhead, we estimate the reduction in the number of hatched eggs could be as high 33% and the reduction in the survival of young fish through the first year as high as 61%. These effects are estimated to be of a magnitude great enough to reduce numbers of these listed salmon species. Approval of the CCC would adversely affect the quality of water to the degree that normal salmon population growth would be severely reduced, and salmon may be extirpated from areas containing cyanide at approved values. These losses would severely diminish the ability of critical habitat to provide for conservation of the southern resident killer whales.

Proposed Critical Habitat of Cook Inlet Beluga Whales

We evaluated the effect of EPA's approval of the cyanide water quality standards on the effect of critical habitat by first reviewing the essential features or primary constituent elements of critical habitat for the proposed designation for Cook Inlet beluga whales. Based on our analysis, the primary features that may be affected by EPA's approved water quality criteria are those primary prey species consisting of Chinook, coho, sockeye, and chum salmon. Based on our analysis, we estimate that coho and Chinook salmon will experience reductions in the number of hatched eggs as high 45% and the reduction in the survival of young fish through the first year as high as 79%. For chum and sockeye salmon, we estimate the reduction in the number of hatched eggs could be as high 37% and the reduction in the survival of young fish through the first year as high as 69%. These effects are estimated to be of a magnitude great enough to reduce numbers of these salmon species. Approval of the CCC would adversely affect the quality of water to the degree that normal salmon population growth would be severely reduced, and salmon may be extirpated from areas containing cyanide at approved values. These losses would severely diminish the ability of critical habitat to provide for conservation of Cook Inlet beluga whales.

Cumulative Effects

Cumulative effects include the effects of future state, tribal, local, or private actions that are reasonably certain to occur in the action area considered in this biological opinion. In this section we focus on the status and trends of land-uses across the United States and the consequences of those land uses for listed and proposed resources. Since our action area encompasses a very broad spatial scale, we focused on key properties of ecosystem condition and the actions that influence those properties. According to the Consultation Handbook (USFWS and NMFS 1998), the "reasonably certain to occur" clause may include such indicators of actions such as approval of an action by a state, tribal or local agencies or government; indications that granting authorities for the action are imminent; project sponsor's assurance that actions will proceed, etc. Although speculative non-federal actions are not factored into the analysis, at the same time "reasonably certain to occur" does not require a guarantee that an action will occur, therefore a degree of uncertainty is acceptable when characterizing cumulative effects.

Due to the scale at which a national consultation occurs, the degree of uncertainty increases, particularly with respect to anticipating the cumulative effects of future non-federal actions across the action area. We necessarily relied on types of human activity (e.g., regional trends and projections in population increases, and associated industrial and commercial development) as proxies for the suite of hydrological, chemical, and biological changes that would reasonably be expected in the surrounding landscape. Metrics of land use (e.g., percent impervious or urbanization; road density) are strongly correlated to a variety of ecological indicators of stress (e.g., changes in aquatic community; increases in chemical constituents, physical stream-channel condition; NRC 2008). Based on our knowledge of past changes within a watershed and the effects landscape changes have had on aquatic ecosystems, we can anticipate the general types and patterns of future land uses will have on the physical, chemical and biological conditions of downstream waterways. The specific factors that are important within a specific locality will vary from place to place, and over time.

The information we present herein is based on data produced by recognized organizations using demographic data, and economic and labor statistics and include their reasoned rough-trend estimates of population and economic change stemming from these data. Changes in the near-term (5-year; 2013) are more likely to occur than longer-term projections (10-year; 2018). Because the anticipated effects are based upon projections that are subject to error and alteration by complex economic and social interactions, our analysis does not address small or localized changes in aquatic habitats. Further, since the effects of future federal actions that are unrelated to the proposed action are not to be considered herein because they require separate consultation pursuant to Section 7, wherever possible, we eliminated known or typical future federal actions from our analysis (e.g., construction of new oil platforms). Many of the actions we discuss herein, such as construction and industrial development, are planned, approved and permitted through wholly local and state approvals and with private funds. However, in many instances we found it impossible to differentiate between non-federal and federal actions, and therefore we erred on including a general type of action in our analysis recognizing that a portion may qualify as federal actions and would not normally be included in our cumulative effects analysis. For example, transportation projects may be undertaken by local and state entities, and others may qualify as federal actions for reasons of federal funding, permitting, etc. In this instance, we were unable to discern federal transportation related actions from future non-federal transportation-related actions therefore we focused on general patterns we might in various regions and the generalized impacts of transportation projects on water quality.

Sources queried for the information herein include the United States Census Bureau, Department of Labor, and Lexis-Nexis information system. With the latter (which was our source for state legislation), we reviewed bills passed in 2007 to 2008 and pending bills under consideration were included as further evidence that actions “are reasonably certain to occur”. Bills that died in process or were vetoed are not included in our review.

Northeast Projection

We began our review for each region by examining current and pending state legislation for regional and local policy and political trends that may impact future development and management directions within the area. For instance, we looked for regulatory and political impetus for changes in zoning, fisheries, environmental standards, and development of commerce and industry. For the Northeast, we selected Maine as a representative state for this effort because of the extent of coastline and waterways, as well as the presence of habitat for several listed species from different taxa. We found that legislation in the state shows tendencies towards controlling invasive species, chemical (wastewater, pesticide, oil, nutrients, bacteria, and other toxic contamination) and sedimentation impacts humans have on rivers and nearshore waters, emissions associated with global warming, and the ability of fish to migrate past river infrastructure. As a general matter, we expect that other coastal states within this region likely have programs or interests engaged in many similar activities, many of which are designed to minimize some of the adverse effects associated with increasing development and extraction industries.

In general, the northeast region is one of the most densely populated regions in the United States. Based upon 2000 United States census data, the northeast United States was predicted to contain 54.8 million people in 2005, and population growth is predicted to decrease over the foreseeable future from 0.41%/year between 2000 and 2010 to 0.24%/year from 2010 to 2020 (USCB 2005a). Much of the regional population is contained in concentrated metropolitan centers. If these cities were to continue to grow at the rate which they did from 2000 to 2007 (USCB 2008), the largest growth will occur in Dover, DE (2.89%/yr), Washington, D.C. metro (1.51%/year), and York-Hanover, PA (1.47%/year). The only population center greater than one million people growing at greater than one percent per year is Washington, D.C. Overall, the northeast United States is predicted to have 55.8 million people in 2010, 56.6 million in 2015, and 57.1 million in 2015. Growth of metropolitan centers will increase discharge of wastewater from water treatment systems into rivers and streams, which will increase the loads of contaminants carried by these waterways to the marine environment, and would have concomitant effects on such parameters as biological oxygen demand, chemical oxygen demand, DO, and water temperature. It is likely that development will continue along the coast and waterways, which will add sediment to river systems and potentially alter spawning habitat. Oil and other roadway pollutants may increase as a result of additional vehicular traffic. Additional recreational use of lakes, waterways, and coastal areas will increase fish takes and add additional discharges from vessels.

Industrial changes can indirectly add pressures to ESA listed species' survival and the health of their habitats. From 2006 to 2016, output of the mining industry is expected to increase by 1.0%/year (Figueroa and Woods 2007), which is a 25% decline in growth from what it was between 1996 and 2006. However, technological advancements will likely increase output in this sector. It should be noted that 60% of this industry is comprised of oil and natural gas, very little of which exists in the northeast United States. Coal output is likely to increase with demand for power through the electrical grid. Most significantly for the northeast, metal mining is anticipated to increase 4.3%/year with demand by various technologies and rising metal process. Currently, granite, peat, roofing slate, iron ore, sulfur, magnetite, manganese, copper, zinc, mica, and precious metals are mined in the region, with numerous others on an infrequent or historical basis (see baseline for additional information). Increasing output by existing and new mines can place additional pressures on species recovery in the foreseeable future by increasing waste runoff into streams and rivers.

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

Output of the transportation industry is expected to increase by 2.9%/year from 2006 to 2016 (Figueroa and Woods 2007), placing additional pollution pressures on listed species and their habitats. Although this rate is slower than the trend from 1996 to 2006, additional movement of freight by truck, plane, and train introduces pollutants, especially oils, to waterways that can increase petroleum concentrations in streams and estuaries. Greater potential for moderate- to large-scale pollutant release by spills and accidents also exists. Carbon dioxide released from petroleum combustion is a significant component of global warming (Vitousek et al. 1997; Nordhaus 2007; EIA 2007) and increases in the transportation will likely mean greater contributions of carbon dioxide and exacerbation of the global warming phenomenon. Based upon these factors, additional recovery pressures are likely to occur from the future growth of the transportation industry.

With increasing population, the leisure and hospitality industry is forecasted to grow by 2.1%/year from 2006 to 2016 (Figueroa and Woods 2007). As with other industries, this is a decline from the 1996 to 2006 rate by about 25%. In addition, most growth will likely occur in food services or drinking places, which is not expected to have impacts to listed species. However, this industry includes personnel and activities that utilize natural and protected areas. Additional use will likely include more debris and pollution discharge into areas frequently used by protected species. It can be contended that additional use of parks can increase outreach and public awareness of protected species and their habitats, which can benefit recovery of these species and areas. It is not known whether growth in the leisure and hospitality industry will have a net positive or negative impact on ESA listed species, but likely will include both helpful and hurtful aspects.

In contrast to other industries, agriculture is forecasted to increase in rate of growth from 2006 to 2016 versus the growth experienced from 1996 to 2006 (Figueroa and Woods 2007). Growth will increase from 1.3%/year to 2.2%/year, a change of roughly 75%. The increase results from increased efficiency from technological improvements and the rise of ethanol from crops. In this sector, agriculture accounts for over 80% of production, which masks regionally important factors. Agriculture in the northeast overshadows a projected output decline in forestry (-0.9%/year) and fisheries/hunting/trapping (-2.9%/year). Agriculture is not as extensive as in other regions of the United States and growth. However, additional growth will increase pollution and sediment runoff into streams, placing additional stress on salmon habitat and making bloom conditions more likely in marine areas where rivers discharge. Based upon the declines in fisheries and forestry, it is unlikely that extensive additional pressures will be placed on ESA listed species recovery by these two industries.

Southeast and Mid-Atlantic Projection

State legislation frequently shows regional and local policy and political trends that can significantly impact future directions within the area. Florida was selected as an example of legislative trends in the mid-Atlantic and Gulf of Mexico because of the extent of coastline, presence of diverse and numerous listed species, socio-economic similarities to other states, large population, and progressive tendencies. Here, legislative regulation is moving towards management of beaches, control of watersheds and vessel discharges, protecting marine resources, restoration of freshwater habitats, identifying issues and contributing factors to climate change, limitation of oil and gas development, and lowering harmful chemical inputs into systems.

Mid-Atlantic states (including Florida) are predicted to increase in population from 55.7 million people in 2005 to 59.8 million in 2010 and 64.0 million in 2015. This is the fastest rate of anticipated regional growth in the nation except for western states (USCB 2005b). The rate of regional growth is anticipated to remain above 10% through 2030 and will be greatest in Florida and North Carolina and lowest in West Virginia. Although this region includes a larger area than the northeast, urban growth is much more extensive in the mid-Atlantic; 12 metropolitan areas experienced population growth of 3%/year or greater from 2000 to 2007, including the Atlanta area, once considered the most rapidly developing area in human history. However, half of these urban centers were in Florida. Cities of over one million people that grew at a rate of 1%/year or greater from 2000 to 2007 included Raleigh, NC (4.49%/year), Atlanta, GA (3.47%/year), Charlotte, NC (3.44%/year), Orlando, FL (3.37%/year), Jacksonville, FL (2.27%/year), Tampa-St. Petersburg, FL (1.96%/year), Richmond, VA (1.51%/year), and Miami, FL (1.16%/year). This rapid and concentrated population increase places much larger demand upon natural systems. Wastewater systems must handle larger loads of sewage. As soil is covered by asphalt and concrete, run-off must be channeled into local stormwater drains increasing contaminant load in streams. Regional areas of development are frequently in low-elevation locations, limiting water retention and movement. Both of these are sources of concern for sediment and contaminants entering local waterways and flowing into rivers, estuaries, and nearshore marine habitats.

Economic development will contribute additional pressures to ESA-listed species of the mid-Atlantic region. West Virginia is mined extensively for coal and demand for this resource to meet the needs of coal-fired power plants will drive increasing production (Figueroa and Woods 2007). Production of North Carolina's cement constituents, Georgian clay, and Florida's phosphate rock are likely to increase with demand in other sectors, such as construction. These and other mining sources can produce excessive sedimentation in streams as well as affect pH and metal concentrations. Expansion or increased production from regional mines is expected to have increased negative impacts to freshwater systems, estuaries, and bay systems in the foreseeable future.

Changes in the leisure and hospitality, transportation, and construction sectors are likely to have similar effects in the mid-Atlantic as were identified for the northeast. However, regional differences will likely lead to different local effects. Low-lying estuaries can collect oil and contaminant run-off from rapidly developing roads, leading to habitat degradation.

The mid-Atlantic region has significantly greater agriculture than in the northeast; a difference that will likely affect the health of streams, estuaries, and marine habitats. Extensive agriculture in the region requires the use of pesticides, fertilizers, and other chemicals in large scale that migrate into freshwater systems. The expansion of agriculture, regardless of crop, will likely entail additional chemicals entering freshwater systems. This can have negative impacts on the survival and recovery of sturgeon populations in fresh water and bay systems, both by accumulation in fish tissues, and general degradation of habitat (i.e., Chesapeake Bay).

West Coast Projection

For the west coast, we selected California as a state representative in legislation. This is because of the large population, complex geography, diverse socio-economic and demographic structure, extent of waterway and coastline, and presence of several listed species of varied taxa. Trends in legislation address the impact and causal regulation of climate change, control of marine debris and harmful substances in waterways and marine areas, regulation of fisheries and invasive species, limitation of oil and gas development, clarification of state listed species takes, and aid for salmon recovery.

States along the Pacific coast, or which contribute water to major river systems here, are projected to have the most rapid growth of any area in the United States within the next few decades. This is particularly true for coastal states and those of the desert southwest. California, Oregon, Washington State, Arizona, Idaho, Utah, Nevada, and Alaska are forecasted to have double digit increases in population growth rates for each decade from 2000 to 2030 (USCB 2005b). New Mexico, Montana, and Wyoming will have far slower growth, with Wyoming forecasted to eventually experience population contraction. Overall, this region had a projected population of 65.6 million people in 2005 and will likely grow to 70.0 million in 2010 and 74.4 million in 2015, making it by far the most populous region (but also containing the greatest land area). As with other regions, growth stems from development of metropolitan areas. However, western growth will come generally from enlargement of smaller cities than from major metropolitan areas. Of the 42 metropolitan areas that experienced 10% growth or greater between 2000 and 2007, only seven have populations greater than one million people. These major cities include Las Vegas, NV (4.79%/year), Phoenix, AZ (4.07%/year), Riverside-San Bernadino-Ontario, CA (3.63%/year), Sacramento-Arden-Arcade-Roseville, CA (2.34%/year), Salt Lake City, UT (1.93%/year), Denver, CO (1.87%/year), and Portland-Vancouver-Beaverton, OR (1.83%/year). It should be noted that most of these metroplexes border coastal or riverine systems. Diffuse, but extensive, growth in the region will increase contaminants from wastewater treatment plants and sediments from sprawling urban and suburban development that enter riverine, estuarine, and marine habitats. This is of particular concern in western states, where numerous rivers and their tributaries are designated critical habitat for listed salmon. Increased contaminant loads have the potential to influence fry and smolt development in freshwater systems. Sediments may alter spawning grounds so as to make them unusable by salmon. Unlike other areas of the United States, the west coast region has extensive fluctuations in elevation and pooling oil and pollutants from developing roadways will likely not be as significant an issue in this region as elsewhere. Western states are widely known for scenic and natural beauty. Increasing resident and tourist use will place additional strain on maintaining the natural state of park and nature areas, also utilized by protected species.

Mining has historically been a major component of western state economies. With national output for metals increasing at 4.3% annually (little oil, but some gas is drawn from western states), output of western mines should increase markedly (Figueroa and Woods 2007). This will increase already significant levels of mining contaminants entering river basins. This future increase is all the more problematic because many western streams feed into or provide spawning habitat for threatened and endangered salmonid populations. These fishes rely upon healthy streams for breeding and their offspring spend the first parts of their lives feeding in rivers, lakes, and streams that heavier contaminant burdens will be affecting. Sturgeon also live in these waterways and will similarly experience negative impacts from growth in the mining sector.

Western states boast large tracts of irrigated agriculture. The rise in agricultural output (Figueroa and Woods 2007) will likely result in two negative impacts upon protected species. With increased production, pesticide, fertilizer, and herbicide use will be used in greater amounts and enter freshwater systems in greater concentrations. Like mining, this has the potential to harm salmonids and sturgeon or their habitats. Further, increased output could place greater demands upon limited water resources. This will reduce flow rates and alter habitat throughout freshwater systems, and likely lead to increased water temperatures and decreases in DO. As water is drawn off, contaminants will become more concentrated in these systems, exacerbating contamination issues in habitats and protected species.

Summary of Cumulative Effects

At the large spatial scale of this consultation, we could not identify specific future state, tribal, local or private actions that were reasonably certain to occur in the action area. Instead we looked at demographic and economic trends to discern general patterns of land use change anticipated by states and federal organizations and their potential effects on listed species. Assuming recent increases in unemployment and poor performance of the dollar are fair indicators of rates potential land use change, regional growth is expected to continue on a slower pace than observed in the past decade. In January 2010, however, unemployment dropped a modest amount from 10 percent to 9.7 percent, which may signal a shift to a more promising economy. However, much uncertainty surrounds whether we will see near term measurable increases in the construction and industrial arenas. We suspect that spatial patterns of growth and development, and redevelopment would likely continue as it has in the past for the near future, but expect that the pace of new development and redevelopment will continue to remain at a slower pace than the past decade.

In general, we expect that the threatened and endangered aquatic species and designated critical habitats considered in this biological opinion are likely to be adversely affected by non-federal activities that affect the quantity, and quality of water, waterways, and habitats important to listed aquatic species and their critical habitat. Non-federal activities that change vegetative cover, soil structure, and water use ways that increase erosion and sedimentation, increase introduction of pollutants into waterways, and result in introductions and spread of non-native invasive species will likely continue to directly and indirectly affect listed species and critical habitats. These species and their critical habitats could also be affected by illegal harvest. At the same time, states or private entities may also engage in activities to restore, enhance, and improve water quality and quantity and restore more natural hydrographic patterns that benefit listed species and their habitats. All of the species and critical habitats considered in this document are likely to be exposed to these types of activities in the future to varying extents.

Integration and Synthesis

The U.S. Environmental Protection Agency proposes to approve state or tribal water quality standards, or federal water quality standards promulgated by EPA, that are identical to or more stringent than EPA's recommended 304(a) aquatic life criteria for cyanide. This approval would authorize states and tribes and EPA to establish source controls (e.g., permits, 401 certifications, waste load allocations, etc.), define and allocate control responsibilities (allocate loads under TMDLs), measure and enforce compliance with the CWA, and measure progress in meeting the goals of the CWA (whether a water body should be listed as impaired; see *Understanding the Water Quality Program* earlier in this Opinion for a summary of the activities that are influenced by or rely upon the water quality standards approved by EPA and implemented by states, tribes and EPA.

In the *Approach to the Assessment* section of this Opinion, NMFS explained that we would assess the effects of EPA's programmatic approval of state, tribal, and federal water quality standards that rely upon their nationally recommended 304(a) aquatic life criteria for cyanide at the CCC and the CMC, by asking:

> Is EPA's approval of state, tribal and federal water quality standards consistent with (or more stringent than) the 304(a) criteria for cyanide, likely to prevent the exposure of endangered species, threatened species, and designated critical habitat to aqueous cyanide concentrations that are toxic, given the approach EPA uses to approve a water quality standards?

If, after considering the best scientific and commercial data available, we conclude that listed resources are not likely to be exposed to activities the water quality standards would authorize, both individually and cumulatively, we stated we would conclude that EPA's proposal to continue recommending the 304(a) aquatic life criteria for cyanide is not likely to jeopardize the continued existence of endangered species, threatened species, or result in the destruction or adverse modification of designated critical habitat under NMFS' jurisdiction. When an agency's national action is likely to prevent exposure of listed resources to their activities, then we would expect an agency's program would generally ensure that actions taken under the program are not likely to individually, or cumulatively, jeopardize the continued existence of threatened and endangered species, and are not likely to result in the destruction or adverse modification of critical habitat that has been designated for those species.

If our assessment determined that listed resources are likely to be exposed to these activities, we stated we would examine whether and to what degree listed species are likely to respond to their exposure, given the approach EPA uses to approve a water quality standards. As part of this analysis, we stated we would examine whether and to what degree EPA has identified chemical, physical and biological scenarios that influence cyanide toxicity and presence in the environment inhabited by listed species and their critical habitat, the nature of any in situ effects, and the consequences of those effects for listed resources under NMFS' jurisdiction, to determine if EPA can insure that the approval of state, tribal and federal water quality standards that they are proposing is not likely to jeopardize the continued existence of endangered species or threatened species, or result in the destruction or adverse modification of critical habitat that has been designated for these species. We stated that we measure risks to listed individuals using changes in the individual's "fitness" or the individual's growth, survival, annual reproductive success, and lifetime reproductive success. When we do not expect listed plans or animals exposed to an action's effects to experience reductions in fitness, we would not expect that action to have adverse consequences on the viability of the populations those individuals represent or the species those population comprise (Mills and Beatty 1979; Stearns 1992; Anderson 2000). As a result, if we conclude that listed plants or animals are not likely to experience reductions in their fitness we would conclude our assessment.

Based on the analysis contained in their BE and on the results of the preliminary screen as introduced by the *Methods Manual*, EPA was able to screen out (or make not likely to adversely affect) determinations on all but 32 species. The 32 species included: several darters, perch, salmonids, and one amphipod. Next, EPA applied a secondary screen that relied primarily on evaluating whether the waters where the 32 listed species occurred were listed as impaired pursuant to the CWA as well as data that would indicate the species had been (1) listed for reasons attributed to cyanide, (2) or whether there were known dischargers of cyanide within the range of the listed species. Using these metrics EPA concluded that of the 32 potentially sensitive species, none would be adversely affected by their action of approving state or tribal water quality standards or federal water quality standards that are equal to or more stringent than the nationally recommended section 304(a) aquatic life water quality criteria for cyanide.

Based on data available in STORET and TRI, as well as information about cyanide in general, the patterns of cyanide exposure are variable and probably not reflective of only permitted discharges. A number of non-permitted (non-point) sources likely also contribute to ambient cyanide concentrations in waters of the United States. Since state, tribal and federal water quality standards form the foundation for, not only permitting, but also evaluating the measuring the progress of the goals of the CWA, it is important to consider non-point sources of a contaminant in evaluating exposure scenarios. Our analysis also demonstrates that permitted discharges likely exceed criterion values from time to time, and can be as much as ten times higher than criterion values without being in violation of CWA. Because we lacked long term data sets for our analysis, we could not evaluate an upper exposure limit nor do we know what a typical exposure scenario would necessarily look like. Our analysis demonstrates that all listed species considered herein would likely be exposed to cyanide during the course of their typical life histories. However, because we could not determine the typical concentrations of exposure, our analysis is premised on the assumption that a suitable concentration for evaluating exposure and response are the proposed criteria values. We believe this is a reasonable threshold for evaluating the effects of cyanide at the national level, since it forms the foundation for a host of water quality management actions in waters of the United States and is the basis for EPA's proposed approval of state, tribal and federal water quality standards.

Our analysis demonstrates that EPA may identify chemical and biological scenarios that influence cyanide toxicity and presence in the environment, but that such information often has little influence (or at least no obvious influence) on the concentration of cyanide that EPA recommends to states and tribes as a "safe dose" for water quality standards. Since the information relegated to "other data" is not considered at the national level in publishing a 304(a) recommendation, then we looked for information to suggest that states would use the information to modify their water quality standards to incorporate site or situation specific modifications as appropriate. That is, we found no evidence that states adopted cyanide water quality standards that were modified by expected water temperatures, unless it was to increase the accepted concentration of the cyanide standard. For instance, since the cyanide standard is driven by rainbow trout data, states with warm water basins often increased the threshold of their water quality standard. In contrast, states where cold water species (e.g., steelhead and salmon) reside did not have modified standards for winter (very cold) water situations that account for the increased toxicity of cyanide at cold temperatures.

In general for cyanide, EPA's decision to recommend and approve water quality standards for cyanide was based on a paucity of data in general, and in particular for listed species. The paucity of data was particularly apparent for saltwater species. However, data was also extremely poor for characterizing a few good case studies on cyanide or what might be considered typical cyanide exposures. Based on our limited review of a few general permits, which incidentally, happen to be one of the most routinely issued permit types issued by EPA and states, generally too few samples are required to result in meaningful monitoring data by which to manage cyanide discharges, or to evaluate the frequency and severity of cyanide entering most basins.

EPA's strict interpretation of what they deemed adequate data for the purposes of decision-making under the CWA is also particularly disconcerting. While both EPA and NMFS are required to use the best available data in their decision-making, when there is data on the listed taxa despite whether there are numerous studies that confirm the findings, NMFS would generally consider that data and the strength of the data in its decision. For instance, EPA often narrowly constrains their decision on a criterion to "avoid confounding factors". However, what might be considered a "confounding factor" in a laboratory setting is often a realistic mixture of conditions in the wild and is relevant for the purposes of evaluating whether a particular action or set of actions is not likely to jeopardize the continued existence of listed resources. For instance, the interplay between DO and cyanide or cyanide and temperature received little attention in EPA's 304(a) aquatic life criteria, despite that there is a wide problem of low DO in many watersheds inhabited by anadromous fish species both on the west coast and the east coast, and salmonids generally inhabit very cold waters during winter months. At least with cyanide, EPA's decision-making process is based on limited very controlled test situations that may be poor predictors of real exposure scenarios and at a minimum, would be strengthened by some field experiments or at least mesocosm studies that are more representative of typical aquatic communities.

Based on our analysis, it also appears that guidance to states and tribes may be prudent for recognizing the potential impacts of cyanide, and the ability of the various forms of cyanide to interact and change within a system. Although we did not search for specific examples of guidance, sources of cyanide within a watershed are numerous and are not limited to expected dischargers and certainly are not limited to the mining industry, which is often the misconception. Based on a review of wastewater treatment facilities, Kavanaugh et al. (2003) caution that managers need to acknowledge that multiple forms of cyanide typically coexist, introconvert, and degrade in a waterbody. It is for this reason, that Kavanaugh et al. (2003) recommended that water quality standards ought to reflect the ability of cyanide compounds to undergo transformation, increasing or decreasing in impact; in so doing, EPA could establish water quality standards for certain classes of cyanide that would be measured using appropriate analytical methods. Kavanaugh et al. (2003) also recommend that the water quality criteria and discharge standards for cyanide be revised to ensure that monitoring methods can distinguish between cyanide forms, and that methods with the greatest potential for use should receive EPA and state approval.

Nevertheless, based upon our analysis we concur with EPA's effect determination that a number species are not likely to be adversely affected when exposed to cyanide at criterion values. Our determination, however, is based on uncertain evidence because for the most part suitable data upon which to make this determination is weak at best. As noted earlier, Gensemer et al. (2007) declined to evaluate the effects of several marine species, acknowledging that the data is too poor to evaluate the protectiveness of the saltwater cyanide criteria on marine species. We concur with Gensemer et al. (2007) that "this represents an area requiring further research" since only three fish genera and five invertebrate genera were used to establish the saltwater criteria. That said, based on the available data as discussed in the preceding analysis, we would not expect the following threatened or endangered species to respond physically, physiologically, or behaviorally to exposure at the CMC or the CCC, whether exposed in saltwater or fresh water or both: Blue whales, bowhead whales, fin whales, humpback whales, North Atlantic right whales, North Pacific right whales, sei whales, sperm whales, beluga whales, southern resident killer whales, Guadalupe fur seals, Hawaiian monk seals, Western Steller sea lions, Eastern Steller sea lions, Florida green sea turtles, Mexico green sea turtles, hawksbill sea turtles, Kemp's ridley sea turtles, loggerhead sea turtles, leatherback sea turtles, Mexico's breeding colonies of olive ridley sea turtles, other olive ridley sea turtles, smalltooth sawfish, elkhorn coral, staghorn coral, white abalone, black abalone and Johnson's seagrass.

Species under NMFS' jurisdiction that demonstrate sensitivity to cyanide at criterion values are: chum salmon, coho salmon, sockeye salmon, Chinook salmon, steelhead, shortnose sturgeon, and green sturgeon, representing 30 DPS/ESUs of these species. Of these species, empirical and modeled evidence suggests that some salmon may die when exposed to cyanide at the CMC of 22 µg/L. According to modeled estimates chum, coho, Chinook, and sockeye salmon, are all more sensitive to cyanide than steelhead, suggesting that some individuals may die when exposed to cyanide at the CMC. However, lethal effects on steelhead salmon are predicated on an exposure to cyanide at low temperatures. That is, the risk of death increases at lower temperatures, while exposure to cyanide in waters at about the average test temperature of 12-13 ºC would probably not lead to the death of steelhead.

While, the relationship between temperature and cyanide may merit further examination to increase confidence in the relationship, existing information suggests that coldwater species may be more sensitive to cyanide at temperatures that are typical of winter months. We have no evidence that the interplay between cyanide and temperature is species specific. If temperature influences the sensitivity of other salmonids, then that would increase the risk of death for not only steelhead, but also coho, sockeye, chum, and Chinook salmon. Our best estimate of effect for steelhead is that roughly 1% of steelhead exposed to cyanide in winter months may die from their exposure, since coho, Chinook, sockeye and chum salmon are all more sensitive to cyanide than steelhead, the percent lethal effect would also increase. We do not know which ages or stages of salmon are most likely to be affected at low temperatures.

Based on our review of chronic studies, we estimate that female sturgeon and Pacific salmon may experience a 40-60% reduction in the number of eggs spawned, and these species would experience a 40 to 70 % reduction in early life stage survival. This should only be considered a rough estimate of the magnitude of the true effect expected at the CCC of 5.2 µg CN/L. Other sublethal responses to low levels of cyanide include reduced swimming performance and reduced weight gain.

In the *Status of the Species* section of this Opinion, we established that Chinook, coho, sockeye, and chum salmon, steelhead, and green and shortnose sturgeon species have declined throughout their range. Some ESUs have demonstrated modest increases in recent years, like Lower Columbia River Chinook salmon and Hood Canal chum salmon, and others like Sacramento winter-run Chinook salmon, Puget Sound steelhead, and Lower Columbia River coho salmon continue to decline. For some ESUs like California coastal Chinook salmon and Central California coast coho salmon, current trends are unknown.

In the *Environmental Baseline* section of this Opinion, we established that salmon and sturgeon are exposed to a myriad of habitat alterations attributable to urban and agricultural development, as well as fishing pressure. Land-use patterns have a profound impact on the contribution of chemicals to the waterways where salmon, steelhead, and sturgeon migrate, rear, spawn, feed and grow. In many basins, these fish are exposed to persistent “legacy” chemicals, as well as there is a relatively constant influx of common-use chemicals like copper and PAHs. At the same time, migratory barriers continue to impact population movement and expansion, loss of riparian forest has lead to increased water temperatures in some areas and the loss of allochthonous input, reduced stream bank complexity, loss of spawning gravels, and altered flow regimes, to name a few. Salmon and sturgeon are also commonly impacted by low DO in many areas throughout their ranges. In the *Cumulative Effects* section of this Opinion, we established that salmon and sturgeon are likely to be exposed to the combined effects of similar habitat modifications for the next ten years, and given expected human population increases and economic development in many regions these impacts will likely increase. The combined effect of these habitat alterations means that chemical loading in many watersheds and coastal areas will likely continue to increase, despite pollution control efforts. Non-point sources for pollutant loading will likely continue to be a significant portion of the problem.

Killing 30-45% of the viable eggs spawned per salmon and sturgeon and killing 56-79% of their larvae is certain to reduce the likelihood of survival and the reproductive success of coho salmon, Chinook salmon, chum salmon, sockeye salmon, green sturgeon, and shortnose sturgeon populations. Reducing the swimming performance of these species would likely reduce their fitness and possibly their survival, through reductions in prey capture, weight gain, displacement, predator escapement, and possibly lead to death. Although there is uncertainty in this analysis, which incidentally is not limited to these calculations, based on the evidence available, we do not believe EPA’s decision-making process mitigates or minimizes these potential losses. Worse yet, EPA and the states are not in a position to detect these losses if or when they occur.

If the intent of the 304(a) aquatic life criteria is to define a level in the waterbody of a pollutant that will be fully protective of the designated uses of a water body and that a state or tribe identify as part of their water quality standards (see BE page 11, and also 40 CFR 131.2), then it would follow that EPA would have to review whether their recommended criteria can protect the specific uses that states and tribes have identified in their designated uses. Instead, our analysis suggests that Gaba (1983) was correct when he noted that EPA and the states are engaged in a water quality process "merely to justify the specific numbers contained in pollutant criteria." That uses are designated without meaningful linkages between the chemical criteria indicators and the biological condition of the waters they are meant to protect, means neither EPA or states or tribes can know how well the chemical criteria are protecting the aquatic assemblages or biological community diversity they are meant to protect. That is, available evidence suggests that EPA (nor states or tribes) is not likely to monitor (a) the direct, indirect, and cumulative impacts of the activities their approvals would authorize on biological community diversity, (b) the nature of those effects on the aquatic assemblages in which they occur, or (c) the consequences of those effects on listed resources. Given the lack of measured endpoints for biological condition, EPA will not know if the aquatic assemblages or species identified as designated are actually protected by the water quality standards, much less whether those water quality standards protect endangered species, threatened species or designated critical habitat under NMFS' jurisdiction.

Based on our review, it is not even clear that EPA would consider listed species as part of the biological community to which Congress directed them to consider in establishing 304(a) aquatic life criteria. EPA's decision-making process (the *Guidelines*) places special emphasis on commercially, recreationally, and other important species, and aquatic assemblages. If, as EPA stated, their only metrics for evaluating the protection of the aquatic assemblage are species richness and species evenness (see EPA 2008a), then EPA could argue (albeit a poor argument) that they are protecting aquatic assemblages if their recommended aquatic life criteria and approved state water quality standards protect non-native aquatic assemblages. Yet, listed species, arguably, are "important" as Congress saw fit to provide for their protection under the ESA and ensure federal agencies have a prominent role in providing for their protection. Moreover, many of NMFS' listed species are also commercially and recreationally valued, and many of the species discussed herein are part of the same aquatic assemblage. Given, EPA's lack of clarity on what constitutes an "important" species, and the indicators they stated they use to evaluate an aquatic assemblage (species richness and species evenness) EPA has placed themselves in a position to exclude the needs of native species in general, and listed species in particular, as part of the biological communities they intend to protect.

All of the endangered species, threatened species, and designated critical habitat under NMFS' jurisdiction depend upon the health of the aquatic ecosystems they occupy for their survival and recovery. EPA's 304(a) aquatic life criteria are designed to reflect the latest scientific knowledge including on the kind and extent of all identified effects on …. fish, shellfish, wildlife, and plants…which may be expected from the presence of pollutants in any body of water…; the concentration and dispersal of pollutants or their byproducts, through biological, physical and chemicals processes; and on the effects of pollutants on biological community diversity, productivity, and stability….. (CWA section 304(a)(1)). As such, 304(a) aquatic life criteria have a prominent role in the success of the overall water quality program designed "to restore and maintain the chemical, physical and biological integrity of the Nation's waters."

Nevertheless, degraded water quality has been one of the contributing factors for the decline of almost all of the anadromous fish species NMFS has listed since the mid-1980s. While cyanide has not been identified as a specific concern in any listing, poor water quality has generally been identified as cause contributing to their need for listing. Generally, it has not been the case that NMFS has isolated poor water quality to only one chemical, physical, or biological stressor for the species that have been listed. To use this lack of evidence, as evidence that an effect is lacking is simply not a persuasive argument that cyanide is not problem for listed species.

Based on our analysis we believe it is reasonable to expect that the number of cyanide sources is likely to increase commensurate with land use changes and expansion of industrial and extraction activities. Our analysis illustrates that the exposure of listed salmon and sturgeon species to cyanide at the proposed chronic criterion concentration is likely to substantially reduce their reproduction by reducing the number of eggs spawned by females, reducing the hatchability of spawned eggs, and by reducing the survivorship of young fish in their first year. These fish may also experience effects on growth, swimming performance, condition, and development. Based upon the magnitude of adverse effects caused by the exposure of these listed species to cyanide at the proposed criteria concentrations, these fish species are likely to become extirpated from waters where they are exposed to approved cyanide discharges that are compliant with approved water quality standards. Continued approval of the EPA's aquatic life criteria for cyanide at the range wide scale of these listed species is likely to reduce their reproduction, numbers, and distribution. Unfortunately, it appears that not only does EPA fail to consider biologically, chemically, and physically relevant exposure scenarios that influence cyanide toxicity, EPA is not and has not put themselves in a position of knowing whether their 304(a) aquatic life recommendations and subsequent approvals of state and tribal water quality standards are in fact, protecting the biological community diversity, productivity and stability they intend to protect. Therefore, we do not believe the EPA can insure that the approval of water quality standards for cyanide are not likely to jeopardize the continued existence of endangered species or threatened species or result in the destruction or adverse modification of critical habitat that has been designated for these species.

Because the proposed action, based on our analysis, is likely to reduce the viability of one or more populations throughout the range of listed Pacific salmon, steelhead, and sturgeon species, we expect that the action is likely to reduce the viability (that is, increase the extinction probability or appreciably reduce their likelihood of both surviving and recovering in the wild) of the listed species as a whole. The specific listed species at risk are: California coastal Chinook salmon, Central Valley spring-run Chinook salmon, Lower Columbia River Chinook salmon, Upper Columbia River spring-run Chinook salmon, Puget Sound Chinook salmon, Sacramento River winter-run Chinook salmon, Snake River fall-run Chinook salmon, Snake River spring/summer-run Chinook salmon, Upper Willamette River Chinook salmon, Columbia River chum salmon, Hood Canal summer-run chum salmon, Central California Coast coho salmon, Lower Columbia River coho salmon, Southern Oregon and Northern California Coast coho salmon, Oregon Coast coho salmon, southern green sturgeon, shortnose sturgeon, Lake Ozette sockeye salmon, Snake River sockeye salmon, Central California Coast steelhead, California Central Valley steelhead, Lower Columbia River steelhead, Middle Columbia River steelhead, Northern California steelhead, Puget Sound steelhead, Snake River steelhead, South-Central California Coast steelhead, Southern California coast steelhead, Upper Columbia river steelhead, and Upper Willamette River steelhead.

Finally, a reduction in Puget Sound Chinook salmon would in turn significantly reduce the forage base of southern-resident killer whales. Therefore, while we agree that southern resident killer whales are not likely to respond physically, physiological, or behaviorally to their direct exposure to cyanide at the CCC or the CMC, we expect that the action, through indirect effects to their primary prey, Pacific salmon, is likely to appreciably reduce the likelihood of southern-resident killer whales surviving and recovering in the wild. Similarly, a reduction in Chinook, coho, sockeye, and chum salmon would in turn significantly reduce the forage base of Cook Inlet beluga whales. We also agree with EPA that Cook Inlet beluga whales are not likely to respond physically, physiological, or behaviorally to their direct exposure to cyanide at the CCC or the CMC, we expect that the action, through indirect effects to their primary prey, Pacific salmon, is likely to appreciably reduce the likelihood of Cook Inlet beluga whales surviving and recovering in the wild.

The proposed action is likely to reduce the habitat qualities for these species that are essential to their conservation. Specifically, reduced availability of clean quality water for the purpose of reproduction, rearing and growth, and a reduction in prey species of sufficient quantity and quality would affect the conservation value of designated critical habitat for these species. The functional value of critical habitat exposed to cyanide at criterion values would be severally reduced and could not serve the intended conservation role for the species. Based on our analysis, the functional value of critical habitat would be reduced throughout the areas designated as critical habitat for: southern resident killer whale, California coastal Chinook salmon, Central Valley spring-run Chinook salmon, Lower Columbia River Chinook salmon, Upper Columbia River spring-run Chinook salmon, Puget Sound Chinook salmon, Sacramento River winter-run Chinook salmon, Snake River fall-run Chinook salmon, Snake River spring/summer-run Chinook salmon, Upper Willamette River Chinook salmon, Columbia River chum salmon, Hood Canal summer-run chum salmon, Central California Coast coho salmon, Lower Columbia River coho salmon, Southern Oregon and Northern California Coast coho salmon, Oregon Coast coho salmon, southern green sturgeon, Lake Ozette sockeye salmon, Snake River sockeye salmon, Central California Coast steelhead, California Central Valley steelhead, Lower Columbia River steelhead, Middle Columbia River steelhead, Northern California steelhead, Snake River steelhead, South-Central California Coast steelhead, Southern California coast steelhead, Upper Columbia river steelhead, and Upper Willamette River steelhead. Similarly, the proposed action would significantly reduce the functional value of proposed critical habitat for Cook Inlet beluga whales when their salmon prey species are exposed to cyanide at criterion values. The result of the exposure of salmon species outside of the geographic area designated as critical habitat would severally reduce the numbers of salmon available to beluga within proposed critical habitat and therefore, the critical habitat could not serve the intended conservation role for the species.

Conclusion

Listed Species and Critical Habitat

After reviewing the current status of the listed species, the environmental baseline for the action area, the effects of the EPA's continuing approval of state water quality standards that rely on their nationally recommended criteria for cyanide and the cumulative effects, it is NMFS' biological opinion that EPA's approval of state water quality standards for cyanide is likely to jeopardize the continued existence of the following species:

> California coastal Chinook salmon, Central Valley spring-run Chinook salmon, Lower Columbia River Chinook salmon, Upper Columbia River spring-run Chinook salmon, Puget Sound Chinook salmon, Sacramento River winter-run Chinook salmon, Snake River fall-run Chinook salmon, Snake River spring/summer-run Chinook salmon, Upper Willamette River Chinook salmon, Columbia River chum salmon, Hood Canal summer-run chum salmon, Central California Coast coho salmon, Lower Columbia River coho salmon, Southern Oregon and Northern California Coast coho salmon, Oregon Coast coho salmon, southern green sturgeon, shortnose sturgeon, Lake Ozette sockeye salmon, Snake River sockeye salmon, Central California Coast steelhead, California Central Valley steelhead, Lower Columbia River steelhead, Middle Columbia River steelhead, Northern California steelhead, Puget Sound steelhead, Snake River steelhead, South-Central California Coast steelhead, Southern California coast steelhead, Upper Columbia river steelhead, Upper Willamette River steelhead, southern resident killer whales, and beluga whales.

After reviewing the current status of the listed species, the environmental baseline for the action area, the effects of the EPA's continuing approval of state water quality standards that rely on their nationally recommended criteria for cyanide and the cumulative effects, it is NMFS' biological opinion that EPA's approval of state water quality standards for cyanide is likely to destroy or adversely modify designated critical habitat for the following species:

> Southern resident killer whale, California coastal Chinook salmon, Central Valley spring-run Chinook salmon, Lower Columbia River Chinook salmon, Upper Columbia River spring-run Chinook salmon, Puget Sound Chinook salmon, Sacramento River winter-run Chinook salmon, Snake River fall-run Chinook salmon, Snake River spring/summer-run Chinook salmon, Upper Willamette River Chinook salmon, Columbia River chum salmon, Hood Canal summer-run chum salmon, Central California Coast coho salmon, Southern Oregon and Northern California Coast coho salmon, Oregon Coast coho salmon, southern green sturgeon, Lake Ozette sockeye salmon, Snake River sockeye salmon, Central California Coast steelhead, California Central Valley steelhead, Lower Columbia River steelhead, Middle Columbia River steelhead, Northern California steelhead, Snake River steelhead, South-Central California Coast steelhead, Southern California coast steelhead, Upper Columbia river steelhead, and Upper Willamette River steelhead.

For species that have no designated critical habitat, then none can be affected.

Species and Critical Habitat Proposed for Listing

After reviewing the current status of bocaccio, canary rockfish, spotted seal, and yelloweye rockfish, the environmental baseline for the action area, the effects of the EPA's continuing approval of state water quality standards that rely on their nationally recommended criteria for cyanide and the cumulative effects, it is NMFS' conference opinion that EPA's approval of state water quality standards for cyanide is not likely to jeopardize the continued existence of bocaccio, canary rockfish, spotted seal, and yelloweye rockfish. NMFS' conclusion for these proposed species is based on the limited data available on marine species. Based on the foregoing analysis, NMFS expects that the approval of cyanide water quality standards is likely to destroy or adversely modify the proposed critical habitat for beluga whales because salmon are an important prey species for beluga whales and are identified as a PCE. NMFS' conclusion for the area designated as proposed critical habitat for Cook Inlet beluga whales is based on the proposed action's effects on salmonids.

Reasonable and Prudent Alternatives

This Opinion has concluded that EPA's approval of state or tribal water quality standards, or federal water quality standards promulgated by EPA for aquatic life criteria that are identical the section 304(a) aquatic life criteria for cyanide, is likely to jeopardize the continued existence of 31 species under NMFS' jurisdiction, and result in the destruction or adverse modification of critical habitat that has been designated for these species. The clause "jeopardize the continued existence of" means "to engage in an action that reasonably would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of listed species in the wild by reducing the reproduction, numbers or distribution of that species (50 CFR §402.02).

Regulations implementing Section 7 of the Act (50 CFR 402.02) define reasonable and prudent alternatives as alternative actions, identified during formal consultation, that: (1) can be implemented in a manner consistent with the intended purpose of the action; (2) can be implemented consistent with the scope of the action agency's legal authority and jurisdiction; (3) are economically and technologically feasible; and (4) would, NMFS believes, avoid the likelihood of jeopardizing the continued existence of listed species or resulting in the destruction or adverse modification of critical habitat.

NMFS reached this conclusion because the evidence available suggests that EPA does not (a) use biological, chemical, or physically relevant information of the natural conditions to which aquatic species would be exposed to derive their numeric recommendations for 304(a) aquatic life criteria or to approve state and tribal water quality standards that rely on their recommended criteria, (b) that EPA is not in a position to know whether the water quality standards they approve actually protect native biological communities, or (c) the listed species that are part of the native biological community. Given the decision structure employed by EPA, EPA will not know whether designated uses are protected, much less whether the direct, indirect, or cumulative impacts of their approval of state and tribal water quality standards that rely on their

304(a) aquatic life criteria recommendations protect endangered species, threatened species, or designated critical under NMFS' jurisdiction.

To satisfy its obligation pursuant to section 7(a)(2) of the ESA of 1973, as amended, EPA must put itself in a position to (a) use biological, chemical, or physically relevant information of the natural conditions to which aquatic species would be exposed to derive their numeric recommendations for 304(a) aquatic life criteria or to approve state and tribal water quality standards that rely on their recommended criteria, (b) monitor whether the water quality standards they approve actually protect native biological communities, and (c) the listed species that are part of the native biological community. What follows is a single reasonable and prudent alternative, consisting of several sub-elements that must be implemented in its entirety to insure that the activities EPA's approval of state and tribal water quality standards would authorize are not likely to jeopardize endangered or threatened species under the jurisdiction of the NMFS or destroy or adversely modify critical habitat that has been designated for these species.

The U.S. Environmental Protection Agency must, by December 1, 2012:

A). Revise the *Guidelines* and any relevant regulatory guidance to:

1. Address how they will incorporate relevant information on biological, chemical, or physical processes that alter a particular chemical's toxicity in nature, in their recommendations such that states and tribes that adopt 304(a) aquatic life criteria as recommended will be required to account for relevant exposure scenarios that affect chemical toxicity, in their state water quality standards.

2. Explicitly address (a) endangered species, threatened species, and designated critical habitat as part of the "important" species the aquatic life criteria are designed to protect, and (b) the native biological community, of which listed species are a part, as the relevant community endpoint to which they intend to protect.

B). Develop and implement the research necessary to replace modeled estimates of species sensitivities to cyanide with direct evidence, using listed species or more closely related surrogates, as the basis for defining cyanide criteria to insure an appropriate level of protection is afforded to listed species and critical habitats addressed by this RPA.

Because this biological opinion has concluded that the U.S. Environmental Protection Agency's proposed approval of state water quality standards that rely on their 304(a) aquatic life criteria is likely to jeopardize the continued existence of endangered and threatened species under the jurisdiction of NMFS, and is likely to result in the destruction or adverse modification of critical habitat, the Environmental Protection Agency is required to notify NMFS of its final decision on the implementation of the reasonable and prudent alternatives.

Incidental Take Statement

Section 9 of the ESA and Federal regulation pursuant to section 4(d) of the ESA prohibits the take of endangered and threatened species, respectively, without special exemption. Take is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct. Harm is further defined by NMFS to include significant habitat modification or degradation that results in death or injury to listed species by significantly impairing essential behavioral patterns, including breeding, feeding, or sheltering. Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. Under the terms of section 7(b)(4) and section 7(o)(2), taking that is incidental to and not intended as part of the agency action is not considered to be prohibited taking under the Act provided that such taking is in compliance with the terms and conditions of this Incidental Take Statement.

Amount or Extent of Take

As described earlier in this Opinion, this NMFS' review of EPA's national approval of state and tribal water quality standards that are consistent with or more stringent than the nationally recommended 304(a) criteria for cyanide. The goal of this national level Opinion is to evaluate the general impacts to NMFS' listed resources from the national approval of the 304(a) cyanide criteria when adopted by states and tribes for implementation as part of their water quality standards. It is not possible to identify take that would occur from specific permitted actions or the specific exposure scenarios typical in a particular state. Instead, this Opinion anticipates the general effects that would occur from the approval of cyanide water quality standards across the landscape. Therefore, this Opinion does not exempt incidental take of listed fish from the prohibitions of section 9 of the ESA for the EPA's approval of cyanide water quality standards.

NMFS anticipates that with implementation of the RPA, incidental take of the listed species considered in this biological opinion is not likely to occur from exposure to cyanide at revised criteria concentrations. However, other elements of water quality standards could allow for exceedance of criteria concentrations and may result in incidental take. The other elements of water quality standards will be the focus of subsequent tiered consultations on individual state and tribal water quality standards. In each of these instances, EPA must conduct a separate, tiered consultation, and if necessary NMFS would issue a separate biological opinion before any endangered or threatened species might be "taken"; the amount or extent of "take" would be identified in those subsequent consultation on site-specific, state or tribal specific, or permit specific activities. Therefore, no incidental take exemptions are provided in this programmatic biological opinion.

Conservation Recommendations

Section 7(a)(1) of the Act directs Federal agencies to utilize their authorities to further the purposes of the Act by carrying out conservation programs for the benefit of endangered and threatened species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on listed species or critical habitat, to help implement recovery plans, or to develop information.

The following conservation recommendations would provide information for future consultation involving EPA's approval of state water quality standards:

1. The EPA should work with states to develop more meaningful linkages between designated uses and the water quality standards they intend to protect, to create monitoring programs that are capable of actually evaluating whether designated uses are being protected by approved water quality standards.

In order to keep NMFS' Endangered Species Division informed of actions minimizing or avoiding adverse effects or benefiting listed species or their habitats, the United States Environmental Protection Agency should notify the Endangered Species Division of any conservation recommendations they implement in their final action.

Reinitiation Notice

This concludes formal consultation on the United States Environmental Protection Agency's approval of water quality standards that are identical to or are more stringent than the section 304(a) cyanide aquatic life criteria. As provided in 50 CFR 402.16, reinitiation of formal consultation is required where discretionary Federal agency involvement or control over the action has been retained (or is authorized by law) and if: (1) the amount or extent of incidental take is exceeded; (2) new information reveals effects of the agency action that may affect listed species or critical habitat in a manner or to an extent not considered in this opinion; (3) the agency action is subsequently modified in a manner that causes an effect to the listed species or critical habitat not considered in this opinion; or (4) a new species is listed or critical habitat designated that may be affected by the action. In instances where the amount or extent of authorized take is exceeded, the United States Environmental Protection Agency must immediately request reinitiation of section 7 consultation.